Habitat reconstruction guidelines for woodland birds: a detailed, focussed, bird-orientated approach

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Contents

Abstract

Habitat reconstruction is needed to reverse severe declines in biodiversity, but opportunities will be limited and many species are facing imminent extinction. Hence, there is a need to ensure reconstructed habitat is successful in every possible opportunity, and this will ultimately depend on the ability of guidelines provided by research to reflect all the habitat requirements of the species concerned. Current assessments of habitat requirements for habitat reconstruction have been successful in identifying a range of important features, but they are based on human-defined sampling using randomly selected plots, transects or patches. While effective at capturing variation in habitat use over broad areas and timeframes, individual samples may not exactly match the scale at which species are operating, and therefore trade-off some of the finer details of habitat requirements.

In this thesis, an alternative, more detailed, focussed, organism-orientated approach was used to determine the important habitat requirements needed to reconstruct habitat for woodland birds in the Mount Lofty Ranges region of South Australia. Specifically, this approach was used to examine the habitat use of woodland birds in an existing system of reconstructed woodland and answer three key questions: 1) Where and how should reconstructed habitat be placed in the landscape, 2) How much habitat needs to be established in these areas, and 3) What microhabitat features should be included?

First, where and how reconstructed habitat should be placed in the landscape was investigated by searching the entire area of habitat for woodland birds in 88 x 1 km^2 cells spread over 160 km^2, to capture species patchily distributed across the landscape. These searches were pooled to examine the influence of 12 landscape features in 22 x 4 km^2 areas on the richness of all woodland bird species and the relative abundance of 19 declining species. The results suggested reconstructed habitat should be established in large blocks along drainage lines and near existing woodland for some hollow users.

Second, how much habitat should be established in these areas was estimated by the total amount of habitat in home ranges to reveal the entire area required by groups of birds. Eight home ranges from three species anticipated to be large area users were determined using radio-telemetry and these estimates were combined with similar data collated from 13 other species studied previously in the same system. The area of habitat used within home ranges ranged from 166 ha to just under 10 ha, suggesting that 100s of hectares would be required to support at least one group of larger area users and that even lower area users may require around 10 ha of habitat to ensure their presence.

Finally, the microhabitat features that should be included were assessed using the fine scale distribution of woodland birds to determine the features that characterise the exact areas of highest use within patches. The distribution of woodland species richness and the richness of declining woodland species were determined by mapping the locations of birds in systematic area searches of five 40-60 ha patches of revegetation, and these were used to guide the sampling of microhabitat features. The findings implied that reconstructed habitat should include a mix of overstorey and understorey plants, comprised of a range of overstorey species, planted at low densities and incorporating a variety of ground substrates.

Overall these results represent a range of important habitat features for woodland birds that can be used to enhance the effectiveness of reconstructed habitat from the landscape down to the microhabitat scale. As these results were developed using a detailed, focussed, bird-orientated approach, they can be used to guide reconstructed habitat with the confidence that they represent some of the finer variation in habitat use. Therefore, together with other results incorporating broader trends, they can be used to increase the chance that any resulting reconstructed habitat will indeed be successful in supporting the species concerned, and ultimately able to ensure their persistence.

Declaration

I certify that this work contains no material which has been accepted for the award of any other degree or diploma in my name, in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text. In addition, I certify that no part of this work will, in the future, be used in a submission in my name, for any other degree or diploma in any university or other tertiary institution without the prior approval of the University of Adelaide and where applicable, any partner institution responsible for the joint-award of this degree.

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This study was also carried out in accordance with the conditions of permits from the University of Adelaide Animal Ethics Committee, the Australian Bird and Bat Banding Scheme, and the South Australian Department for Environment and Heritage.

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Chapter 1
General Introduction

1.1 Background

The clearance of vegetation associated with human expansion has been immense and has resulted in severe declines in biodiversity, to the extent that habitat loss is ranked as the number one factor causing species decline throughout the world (Vié *et al.* 2009). In response, substantial efforts have been made to protect and restore the habitat that remains. For example, in Australia the broad scale clearance of native vegetation has been stopped under various legislative acts in most states and territories (e.g. South Australian Native Vegetation Act 1991), and a national system of reserves has been established - many of which are actively managed for the primary purpose of maintaining biodiversity (Commonwealth of Australia 2005). However, despite these efforts the declines in biodiversity are ongoing with many species continuing to disappear from certain locations and regions (Recher 1999, Ford *et al.* 2001, Ford 2011). Moreover, these continued declines cannot be attributed to the degradation of the remaining habitat alone, and instead appear to be associated with an extinction debt caused by past vegetation clearance and the limited amount of habitat that remains (e.g. MacHunter *et al.* 2006, Ford *et al.* 2009). Therefore, it has been widely recognised that protecting and restoring the remaining habitat will not be enough on its own and substantial amounts of habitat will need to be reconstructed on cleared land if biodiversity is to be conserved (Saunders & Hobbs 1995, Recher 1999, Vesk & Mac Nally 2006).

Habitat reconstruction however, faces significant challenges. For instance, many species have declined to extremely low levels and face imminent extinction over the next few decades if suitable habitat is not reconstructed (Recher 1999). Hence, there is a need to ensure habitat reconstruction is successful, as there are unlikely to be any second chances. Furthermore, the opportunities for habitat reconstruction are likely to be limited, as revegetation is expensive (Schirmer & Field 2002) and the land required will be difficult to obtain because land reconstructed is lost to agricultural production

(Vesk & Mac Nally 2006), and current revegetation patterns suggest most farmers are unwilling to give up large areas of productive land (Bennett & Mac Nally 2004). Hence, the pressure on habitat reconstruction to be successful in every possible opportunity is enormous, and we need to be absolutely confident that the habitat reconstructed will be a success.

Ensuring habitat reconstruction is successful will necessarily require an understanding of what constitutes habitat for the species concerned. However, habitat is a complex phenomenon that is species specific and occurs over a range of spatial scales, from biogeographic regions through to foraging and nesting sites (Johnson 1980, Hutto 1985, Wiens *et al.* 1986, Wiens *et al.* 1987). To provide management recommendations on the conservation of species within biogeographic regions though, three scales are typically investigated: the landscape, local area (i.e. habitat patches or territories), and microhabitat (e.g. Saab 1999, Luck 2002, Oppel *et al.* 2004, Barbaro *et al.* 2008); which may be biologically significant for a number of reasons. For example, metapopulation theory suggests any given species requires appropriate landscape features, such as particular extents and configurations of habitat to support viable populations (Hanski *et al.* 1996, Hanski 1999, 2001); while physiological and morphological traits of species such as body size and diet, suggest particular sized areas of relevant habitat will be required within landscapes to sustain individuals and groups (McNab 1963, Schoener 1968, Harestad & Bunnel 1979); and the concepts of niche and resource partitioning suggest specific microhabitat features will be required within these areas to ensure the survival of species in the face of limited resources and competition (Cody 1974, Schoener 1974, 1982). Similarly, a vast body of empirical research has also highlighted the importance of these scales, with the presence of species in landscapes linked to thresholds in habitat extent and particular configurations (e.g. Andren 1994, Radford & Bennett 2004, Radford *et al.* 2005); the use of areas by species within landscapes associated with specific patch areas (e.g. Helzer & Jelinski 1999, Shake *et al.* 2012) or amounts of habitat in territories (e.g. Carey *et al.* 1990, Wiktander *et al.* 2001); and the behaviour of species within patches closely tied to particular plant species (e.g. Holmes & Robinson 1981, Recher & Majer 1994) or substrates (e.g. Holmes *et al.* 1979, Recher 1989). Clearly, features across all of these scales are critical components of habitat for species, and therefore ensuring habitat reconstruction is successful will ultimately depend on the ability of guidelines provided by research to closely reflect all of these requirements.

There has already been a large amount of research dedicated to determining important habitat requirements and using these to develop guidelines for reconstructed habitat (Table 1.1). These studies have identified a range of important features and formed valuable guidelines on the landscape, area and microhabitat requirements for a range of different species and taxa. It is possible though, that a different level of understanding could be gained and even more details of habitat requirements may still yet be uncovered. For instance, nearly all of this research has been performed using samples that are human-defined in their size, shape and placement; e.g. randomly selected quadrats, transects, points or patches are used to record species use and assess habitat requirements (Table 1.1). In contrast, far fewer studies directed at determining habitat requirements for reconstructed habitat have used the locations of individual organisms to assess habitat requirements, or explicitly catered for their distribution in sampling designs (e.g. Gabbe *et al.* 2002, Shanahan *et al.* 2011b).

Table 1.1. List of studies that have developed guidelines for habitat reconstruction, along with the taxa studied, the habitat requirements assessed and the sampling method employed. Continued over page.

Study	Taxa	Habitat requirements assessed			Sampling method
		Landscape	Area	Microhabitat	
Barrett & Davidson (1999)	Birds	✓	✓	✓	Quadrat
Freudenberger (2001)	Birds	✓		✓	Quadrat
Major *et al.* (2001)	Birds	✓	✓		Transect
Watson *et al.* (2001)	Birds		✓		Patch
Brooker (2002)	Birds		✓		Patch
Mac Nally & Horrocks (2002)	Birds		✓	✓	Transect
Twedt *et al.* (2002)	Birds			✓	Quadrat
Arnold (2003)	Birds			✓	Quadrat
Westphal *et al.* (2003)	Birds	✓			Point
Huggett *et al.* (2004)	Birds		✓		Patch

Table 1.1. Continued.

Study	Taxa	Habitat requirements assessed			Sampling method
		Landscape	Area	Microhabitat	
Law & Chidel (2006)	Bats	✓	✓	✓	Patch
Cunningham *et al.* (2007)	Rept/Mam*	✓	✓		Transect
Kavanagh *et al.* (2007)	Birds		✓		Point
Loyn *et al.* (2007)	Birds	✓	✓	✓	Quadrat
Maron (2007)	Birds		✓	✓	Transect
Thomson *et al.* (2007)	Birds	✓			Quadrat
Westphal *et al.* (2007)	Birds	✓			Point
Barrett *et al.* (2008)	Birds		✓	✓	Transect
Selwood *et al.* (2009)	Birds	✓	✓	✓	Quadrat
Thomson *et al.* (2009)	Birds	✓			Quadrat
Lindenmayer *et al.* (2010)	Birds	✓	✓	✓	Point
Mac Nally *et al.* (2010)	Birds		✓	✓	Patch
Twedt *et al.* (2010)	Birds	✓		✓	Quadrat
Gardali & Holmes (2011)	Birds	✓		✓	Point
Law *et al.* (2011)	Bats	✓	✓	✓	Patch
Munro *et al.* (2011)	Birds	✓	✓	✓	Point
Shanahan *et al.* (2011a)	Birds		✓	✓	Transect
Yen *et al.* (2011)	Birds			✓	Transect
Lindenmayer *et al.* (2012)	Birds			✓	Transect
Polyakov *et al.* (2013)	Birds	✓			Transect
Freeman *et al.* (2015)	Birds	✓			Quadrat
Gould & Mackey (2015)	Birds	✓		✓	Quadrat
Smith *et al.* (2015)	Vertebrates			✓	Quadrat

* Reptiles/Mammals

The human-defined approach has clearly been effective at determining habitat requirements, and has distinct advantages in that it is easily replicable with upwards of 100 samples often employed over areas greater than 100 km^2 in size and repeated over multiple years (e.g. Brooker 2002, Loyn *et al.* 2007, Yen *et al.* 2011). However, there may be a trade-off for this spatial and temporal scope in capturing some of the finer variation in habitat use, as individual samples may not exactly match the scale at which species are operating. This may be the case, as species can display significant spatial variation in habitat use in response to natural heterogeneity in habitat over all spatial scales. For example, home range and spot mapping studies have shown that even within individual vegetation types, species only occupy specific areas in relation to finer scale differences in habitat (e.g. Wiens 1985, Misenhelter & Rotenberry 2000, Luck 2002, Furey & Burhans 2006), indicating that they are likely to vary over large areas such as landscapes (e.g. Fig. 1.1). Furthermore, studies examining the areas used by individuals or groups have demonstrated that these occupied areas can span multiple patches if the habitat is fragmented (Andren 1994), and therefore that their distribution will not necessarily correspond to a single patch (e.g. Fig. 1.2). Moreover, home range and territory studies have also revealed that individuals and groups do not use the whole of the areas they occupy equally, and demonstrate core areas of use according to the distribution of specific microhabitat features (e.g. Chamberlain & Leopold 2000, Barg *et al.* 2006, Anich *et al.* 2012, Broughton *et al.* 2014; Fig. 1.3).

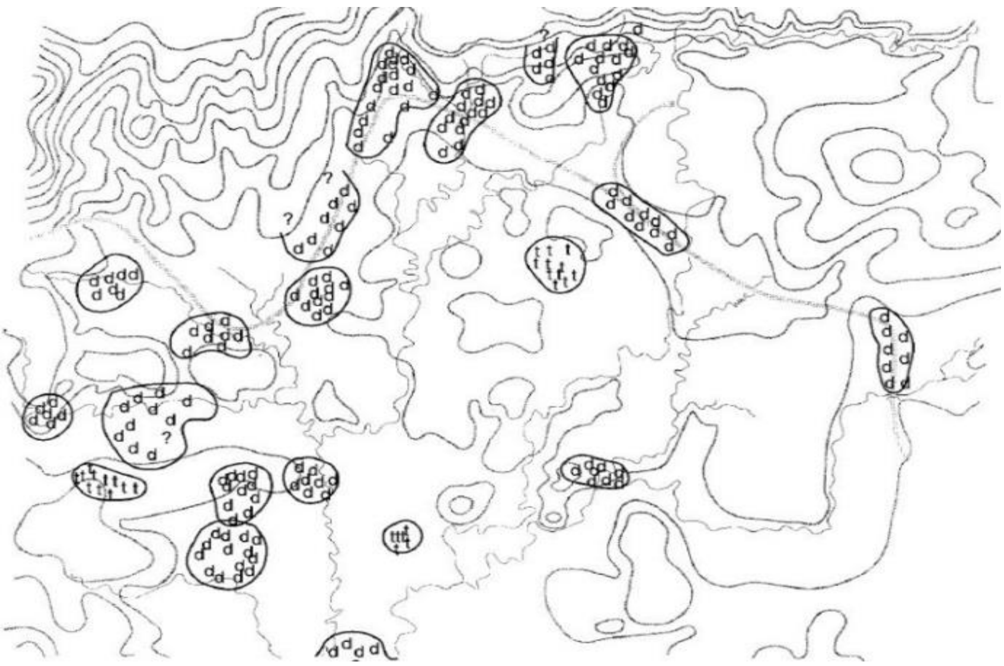


Fig.1.1. Spot maps of two bird species obtained over a 104 ha area of tropical forest indicating spatial variation in habitat use at the landscape scale. d = Dusky Antbird (*Cercomacra tyrannina*) and t = Long-tailed Tyrant (*Colonia colonus*). The area was censused weekly from January to July over a two year period by walking parallel transects spaced 100 m apart. The location of each letter indicates at least one census registration. Both species were primarily associated with gaps in forest canopy. NB. Thick lines represent contours 20 m apart, while thin lines reflect streams and the hatched grey line corresponds to a road. From Robinson *et al.* (2000).

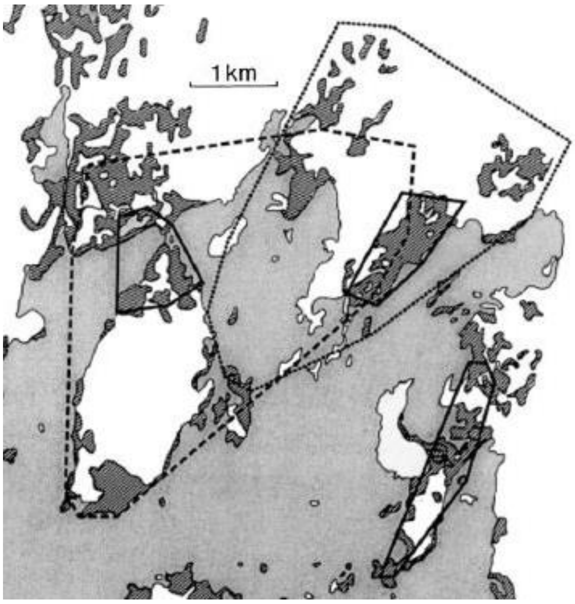


Fig. 1.2. Home ranges of the Lesser Spotted Woodpecker (*Dendrocopos minor*) in southern Sweden, demonstrating differential use of habitat patches. Dark hatched areas represent the old deciduous forest preferred by this species, while grey indicates water, and white areas represent coniferous forest or open agricultural land. The polygons show the winter home ranges of two males in the same year (broken lines) surrounding their late spring breeding territory (continuous lines), and the late spring breeding territory of one female. Adapted from Wiktander *et al.* (2001).

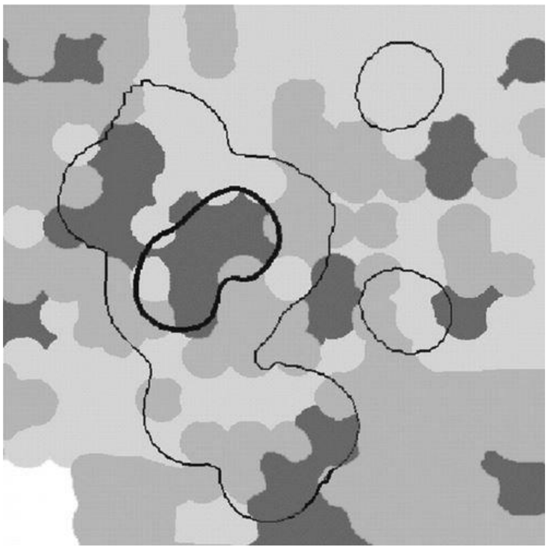


Fig. 1.3. Within-territory distribution of a male Cerulean Warbler (*Dendroica cerulea*) and the associated canopy tree species distribution, illustrating spatial variation in habitat use at the microhabitat scale. The 95% kernel territory boundary is indicated by the thin black line and the core area by the heavier line. At the territory level this male and six others used tree species in proportion to availability, but within the territory core areas were found to be associated with bitternut hickory (dark grey areas) which were used as a song posts and were thought to have a foliage architecture that facilitated song transmission. White areas are ash canopy trees, medium grey areas are sugar maple, and light grey areas are all other species. From Barg *et al.* (2006).

Given the extent of this variation, it is inevitable that in any given sampling regime some samples will not coincide with the exact distribution of habitat use and some of the finer variation in habitat use may be missed. For example, Barg *et al.* (2006) found that sampling using the variation displayed within territories, highlighted the most important microhabitat features from those already gained without using this variation at the territory level (see Fig. 1.3). Based on the other examples provided, similar scenarios can also be envisaged at the landscape and area scales, e.g. a small randomly placed plot in Fig.1.1 may only detect and infer the importance of surrounding landscape features for one of the species but a wider sample would more likely highlight the importance of the landscape to both, and considering all the habitat patches used home ranges in Fig. 1.2 would reveal more of the area used than considering only one patch. As the range of features identified by previous studies demonstrates though, capturing these extra details may not be a problem for determining the more major or obvious requirements. However, if more of this variation could be captured in an approach that is able to closely reflect habitat use, then it may help to elucidate some more subtle or cryptic requirements.

Capturing these extra details however, will be difficult and would require a more intensive sampling effort than the traditional human-defined approach. For instance, surveying only a small area of land in Fig. 1.1 would require far less effort than searching the whole area. Similarly, tracking birds and documenting all the areas used in the other two examples would be much more intensive than simply recording their use of individual patches or plots. Hence, capturing the extra details of habitat use, whilst maintaining the spatial and temporal scope of previous research, would be infeasible, if not impossible, and therefore, an alternative more focussed approach would be required (*sensu* Wiens 1989). This may invariably sacrifice the ability to capture some inter-regional or longer-term trends, but as evidenced by the preceding examples, may provide further important insights into habitat requirements. In terms of developing guidelines for habitat reconstruction this may be invaluable, as any extra detail on what constitutes habitat will help to create the best possible habitat in the limited opportunities provided, or at the very least confirm current findings, and therefore increase the confidence that reconstructed habitat will be a success.

1.2 Study aims

In this thesis, a more detailed, focussed, organism-orientated approach designed to closely reflect the use of habitat was applied to the problem of developing guidelines to enhance future reconstructed habitat for woodland birds in the Mount Lofty Ranges region of South Australia. Since European settlement southern Australia has been especially hard hit by vegetation clearance with many regions having lost around 90% of their pre-European habitat (NLWRA 2001). In particular, the woodland systems associated with better quality agricultural land on lower elevations and deeper soils have been disproportionately cleared (NLWRA 2001), and this has led to severe declines in woodland birds (Ford *et al.* 2001, Ford 2011). The Mount Lofty Ranges region epitomises these changes with only 7% of the original vegetation remaining, of which only 2% remains at lower elevations where most of the woodland occurred (Paton *et al.* 1999, Paton *et al.* 2004). Already 8-10 woodland bird species have disappeared from the region and despite the cessation of habitat clearance, around 50 more are continuing to decline in distribution and abundance (Paton *et al.* 1999, Paton *et al.* 2004, Szabo *et al.* 2011). Habitat reconstruction is desperately needed to halt these declines (Paton *et al.* 2004, Szabo *et al.* 2011) and ensuring its success will be vital if further losses are to be avoided.

Specifically, the aim of this thesis was to answer three key questions corresponding to the three major scales of habitat requirements: 1) Where should reconstructed habitat be placed in the landscape in order to support a range of typical woodland and declining woodland bird species, 2) How much habitat should be placed in these areas in order to support individuals and groups of these species, and 3) What are the key microhabitat features that should be included in these areas to ensure they provide the specific resources required by these individuals and groups? These three questions formed the three core chapters of this thesis.

1.3 Study area

In order to answer these questions, the habitat requirements of woodland birds were studied in an existing system of reconstructed woodland at Monarto about 60 km east of Adelaide on the eastern edge of the Mount Lofty Ranges (Fig. 1.4). The reconstructed woodland was planted in the mid to late 1970s to reduce the effects of dust and erosion, and improve the aesthetics of the area in order to pave the way for a satellite city to Adelaide that was later cancelled (Paton *et al.* 2004). Before plans were abandoned though, 1850 ha of cleared agricultural land was revegetated, more than doubling the vegetation cover in the region (Paton *et al.* 2010b). Around 600, 000 plants were established comprising about 250 species of trees and large shrubs originating from all around Australia and some from overseas, and today the area resembles open woodland (Paton *et al.* 2004, Paton *et al.* 2010b).

This system was chosen to conduct this study for a number of reasons. First, Monarto is typical of many woodland systems throughout the Mount Lofty Ranges and southern Australia, as it is situated at low elevations (< 250 m above sea level; Department of Environment Water & Natural Resources), receives a moderate level of rainfall of around 400 mm annually (Bureau of Meteorology 2016), and formerly had much of its area covered by woodland of which nearly 90% has now been cleared (pre-European vegetation mapping, Department of Environment Water & Natural Resources). Therefore, any results from this system should be broadly applicable to similar woodland areas.

Second, the reconstructed woodland at Monarto has a range of characteristics that make it particularly suitable for answering the research questions. For instance, the broad scale of the plantings means they vary in regard to landscape attributes such as their proximity to remnant vegetation and association with topographic features like drainage lines, while the size of the plantings also means they are theoretically large enough to support individuals and groups from species with a range of different area requirements, and at the microhabitat scale, the range of plant species established means there is also considerable variation in their structure and floristics. Assessments of requirements at each of these scales should therefore be comprehensive and results robust.

Finally and most importantly, this system was chosen because unlike most other revegetated areas the reconstructed woodland at Monarto has provided habitat for a range of woodland bird species, and at least for woodland birds is a unique example of successful reconstructed habitat. For example, most revegetated areas are limited in their habitat value for woodland birds as they are mainly used by more common or generalist bird species (Harris 1999, Kimber *et al.* 1999, Ryan 1999), do not support certain functional groups (e.g. bark foragers (Martin *et al.* 2004, Loyn *et al.* 2007); or ground foraging insectivores (Barrett *et al.* 2008)), and generally fail to match the levels of richness found in remnant vegetation (Munro *et al.* 2007, Martin *et al.* 2011). In contrast, the reconstructed woodland at Monarto has provided habitat for a wide range of woodland birds with 89% of the woodland bird species present within the wider Mount Lofty Ranges recorded using the plantings (Paton *et al.* 2010a), including many species listed as declining in the rest of the region and southern Australia (Leary 1995, Paton *et al.* 2004). Determining the habitat features responsible for this success will therefore be invaluable for guiding future habitat reconstruction.

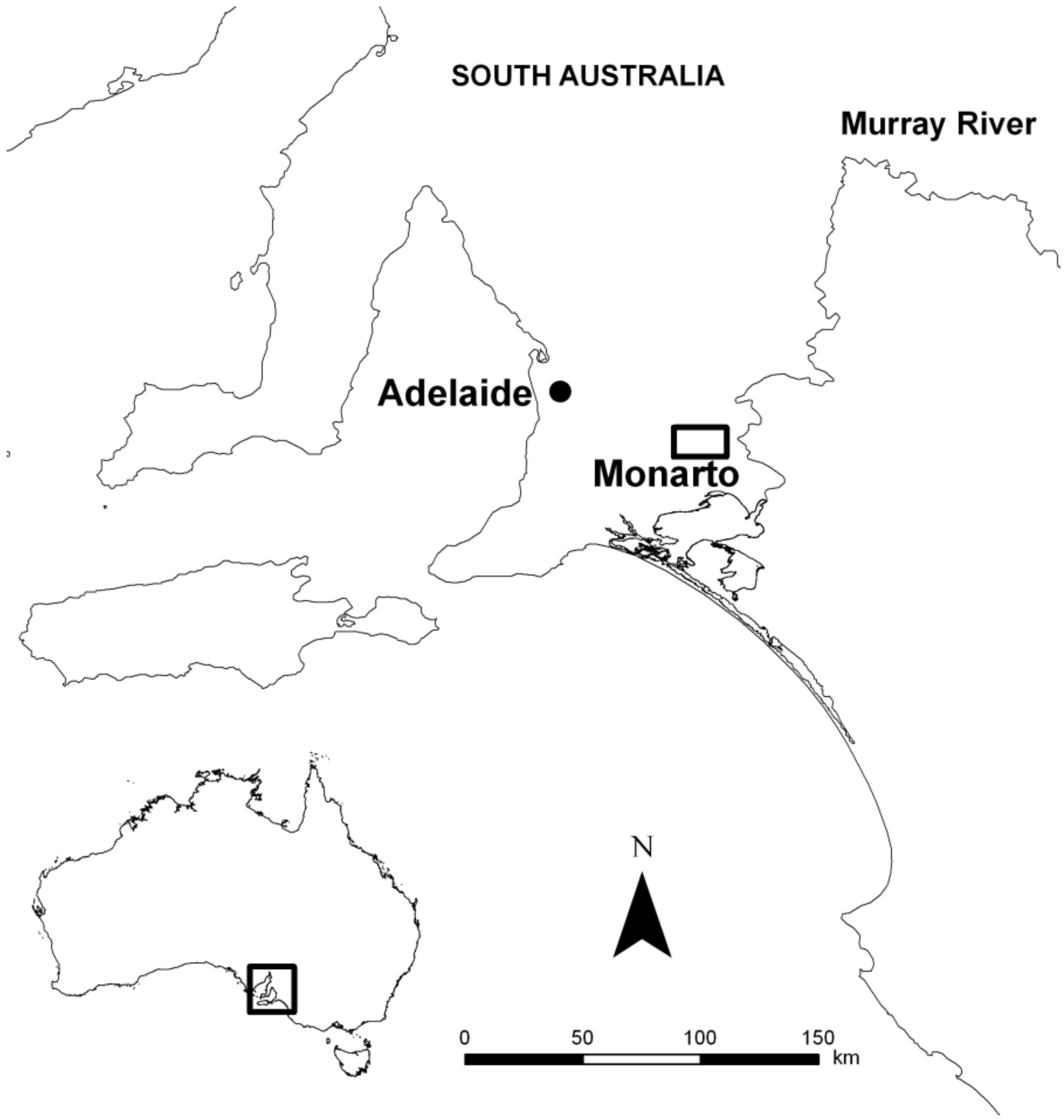


Fig. 1.4. Location of the Monarto region within South Australia.

Chapter 2

Determining where and how reconstructed habitat should be placed using landscape scale sampling of woodland birds

2.1 Abstract

Extensive habitat reconstruction is required to counteract losses in biodiversity, but reconstructing large areas will be expensive and therefore the land able to be reconstructed will be limited. Hence, there will be a need to prioritise where and how reconstructed habitat is placed to obtain the best outcomes from the funding available, and this will require knowledge of the landscape features that most influence biodiversity. Existing research has identified a range of important landscape features to guide reconstructed habitat, however these have all been determined by sampling biodiversity in only small portions of landscapes. This technique is easy to employ over broad areas and may not be an issue for most species, but for some patchily distributed species may not capture all the relevant variation. To provide a robust assessment of landscape requirements for these species, in this study landscape features were assessed using landscape scale sampling in order to guide the placement of reconstructed habitat for woodland birds in the Mount Lofty Ranges region of South Australia. Area searches of 88 x 1 km^2 cells were used to sample woodland birds in an existing system of reconstructed woodland spread over a 160 km^2 region. These were pooled to assess the relative importance of 12 landscape features in 22 x 4 km^2 areas on woodland bird species richness and the relative abundances of 19 declining species. The length of drainage lines associated with the revegetation was the most influential feature followed by the total area of revegetation for woodland species and most declining species, while the size and shape of the plantings were also important for some individual species. Overall, remnant vegetation was unimportant, but the area of woodland remnant was influential for two species, both of which use hollows – a feature currently missing in the revegetation. These results reinforce existing findings obtained using smaller samples over broad scales, and together suggest reconstructed habitat should be placed

in large blocks associated with drainage lines and also near existing woodland for certain species. As a result, these features can be used to guide the placement of reconstructed habitat with the knowledge that they are in fact important features for both patchily distributed and more widespread species, and therefore that biodiversity outcomes will indeed be maximised from the funding available.

2.2 Introduction

It is widely acknowledged that the reconstruction of habitat over broad-scales is required to counteract severe losses in biodiversity associated with widespread vegetation clearance (Saunders & Hobbs 1995, Recher 1999, Vesk & Mac Nally 2006). Broad-scale reconstruction of habitat however, will be expensive as the cost of revegetation can be into the thousands of dollars per hectare (Schirmer & Field 2002), meaning millions will be required to revegetate the extensive areas required. Sourcing such large amounts of funding is likely to be difficult, and as a result the land able to be reconstructed will be limited. Hence, there is a need to prioritise the placement of reconstructed habitat in order to achieve the best possible biodiversity outcomes from the funding available (Bennett & Mac Nally 2004, Vesk & Mac Nally 2006, Thomson *et al.* 2007).

Prioritising the placement of reconstructed habitat will require an understanding of the landscape requirements of species, as many species have a range of traits that are directly affected by landscape level habitat loss and fragmentation that can influence their ability to survive. For example, species that are rare, sedentary or specialised in their habitat requirements may be vulnerable to different levels of landscape fragmentation due to reduced ability to maintain viable population sizes, move between patches and exploit available habitats (Wiens 1995, Mac Nally 1997, Mac Nally *et al.* 2000a), and may therefore require particular landscape features to survive. Indeed, the presence of species in landscapes has been linked to a vast range of landscape features, most of which relate to the extent of habitat (i.e. overall amount of suitable habitat), configuration of habitat (i.e. the size, shape and aggregation of habitat patches), or the composition of habitat in the landscape (i.e. the proportion of different types of habitat;

Bennett *et al.* 2006). For the purposes of reconstructing habitat, the extent and composition can inform 'where' habitat is placed in the landscape (i.e. in landscapes with a certain amount of habitat or specific types of habitat), and configuration can guide 'how' habitat is placed (i.e. in certain sized or shaped patches and aggregations). Hence, understanding which particular features in these categories are important for the species in question will be essential in effectively prioritising habitat reconstruction and maximising outcomes from the funding available.

The problem of determining influential landscape features to guide reconstructed habitat has already received some attention with a range of studies assessing the effect of different features on fauna (e.g. Westphal *et al.* 2003, Law & Chidel 2006, Cunningham *et al.* 2007, Kavanagh *et al.* 2007, Selwood *et al.* 2009, Lindenmayer *et al.* 2010, Mac Nally *et al.* 2010, Twedt *et al.* 2010, Gardali & Holmes 2011), and several also using these assessments to develop predictive spatial models for the placement of habitat in certain regions (e.g. Thomson *et al.* 2007, Westphal *et al.* 2007, Mac Nally 2008, Thomson *et al.* 2009). These studies have provided valuable guidelines for the placement of reconstructed habitat for a range of species in a range of different regions. However, in all these cases, faunal use (e.g. species richness, incidence) was recorded in only small parts of the landscape. For instance, fauna are typically recorded in small patches, or in plots (quadrats/transects) usually 1-2 ha in size, which are used to represent landscapes that may contain 10s or 100s of hectares of habitat. This technique has clearly been effective, and may not be an issue for the majority of species which are widespread and have generalist requirements, but for rare species with specific habitat requirements that are patchily distributed, it may not capture all the relevant variation (Robinson *et al.* 2000). For example, research on home range and territory selection has shown that even within individual vegetation or habitat types some species only occupy specific areas according to finer scale differences in habitat (e.g. Wiens 1985, Misenhelter & Rotenberry 2000, Luck 2002, Furey & Burhans 2006). Therefore, if only a very small portion of the habitat is sampled it is likely that at least some samples will fall in unused areas and such species may never be detected. Indeed, despite considerable replication, many studies cite a number of species that were recorded incidentally in a study area but never in a sample (see Watson 2003 for examples), while others have found the number of species detected in small areas can be markedly less than those recorded in larger areas of the same habitat (e.g. Mac Nally 1997, Robinson *et al.* 2000, Watson 2004). Hence, for some patchily distributed species small samples may not provide the best assessment of landscape features. This may be significant for reconstructing habitat, because many of the target species are declining and no longer widespread or common, and have specific microhabitat requirements (e.g. hollows; Gibbons & Lindenmayer 2002, or fallen timber; Antos *et al.* 2008) that are unlikely to be homogeneously distributed in space.

A logical solution to ensure patchily distributed species are not missed is to sample or 'area search' the entire area of habitat (*sensu* Watson 2003). Area searches have been used to sample individual patches of habitat (e.g. Brooker 2002), but to our knowledge have not yet been used to sample across multiple patches at the landscape scale. Searching the entire area of habitat at this scale would undoubtedly require much more effort than surveying only a small portion of the area, and therefore the ability to replicate over multiple landscapes and detect broader trends may be reduced. At the patch level though, area searches have been demonstrated to be much more complete in terms of the species detected than small samples like plots and transects (Watson 2004). Therefore, while intensive, area searches may represent an important complementary technique that is able to more consistently detect patchily distributed species and provide a robust assessment of their landscape requirements.

Here, landscape scale area searches were used to ascertain the landscape features that should guide the placement of reconstructed habitat for a range of woodland birds in the Mount Lofty Ranges, South Australia. The Mount Lofty Ranges, like other regions in southern Australia has lost a substantial portion of vegetation (> 90%), and in particular woodland has been disproportionately cleared, which has led to major declines in woodland birds (Paton *et al.* 1999, Paton *et al.* 2004, Szabo *et al.* 2011). To counteract the effects of vegetation loss, a goal has been set to increase the extent of functional ecosystems to 30% of the region by 2028 (AMLR NRM Board 2014), which will necessarily involve reconstructing large areas of woodland. This presents an opportunity to reverse the declines of woodland birds, and prioritising where and how this habitat should be placed will be vital if the most is to be made of this chance.

In order to effectively guide the placement of reconstructed habitat in the region, the aim was to establish the following: 1) does reconstructed habitat need to be in close proximity to other plantings or is it just the overall area in the landscape that is important, 2) does it need to be planted as single large patches or can it be in multiple smaller patches, 3) should it be planted in large blocks or can it be in narrow strips, 4) does it need to be placed near remnant vegetation, and if so does this need to be woodland or can other remnant types also be beneficial, 5) is there benefit in planting in potentially more productive areas of the landscape, and 6) does planting in association with a variety of habitat types increase the value?

To answer these questions, variation in corresponding landscape features was compared to the variation in woodland birds using different parts of an existing system of reconstructed habitat. The number of woodland species using the plantings was used to assess the associated landscape features, along with the relative abundances of a range of declining woodland birds.

2.3 Methods

2.3.1 Study area

The study was conducted in an area surrounding the town of Monarto, approximately 60 km east of Adelaide on the eastern plains of the Mount Lofty Ranges (35°3'S, 139°2'E - 35°9'S, 139°13'E). In the 1970s, this area was to be the site of a satellite city to Adelaide that was later cancelled, but as part of the plans 1850 ha of woodland was planted to ameliorate the threats of dust and erosion, and improve the aesthetic character of the site (Monarto Development Commission, unpublished report). The woodland was all of similar age (1974-1979), planted using the same method (rows of tubestock 4-6m apart), and with similar floristic composition (Paton *et al.* 2004), yet it was situated on several different landforms, at varying distances from remnant vegetation, and in a range of sizes and shapes. As a result, this system was an ideal setting for determining the influence of landscape features on reconstructed habitat.

2.3.2 Study design

To capture the variation in woodland birds, bird surveys were conducted over a grid of 160 x 1 km^2 cells spanning the extent of the 1970s plantings. Both revegetation and remnant vegetation were surveyed, although for the purposes of this study only the data collected in the revegetation was considered. To reduce the influence of boundary effects and better represent the variation in landscape features, this grid was later converted to a set of 22 x 4 km^2 cells (Fig. 2.1). These 22 cells were only those that overlapped the revegetation, did not display any remaining boundary effects and had the majority of their revegetation area surveyed (average of > 75% across both survey periods – see below). The 4 km^2 cell area also incorporated the largest home range area recorded for woodland birds of 2.5 km^2 (Chapter 3), which theoretically allowed most species to be present in a cell if landscape features were appropriate.

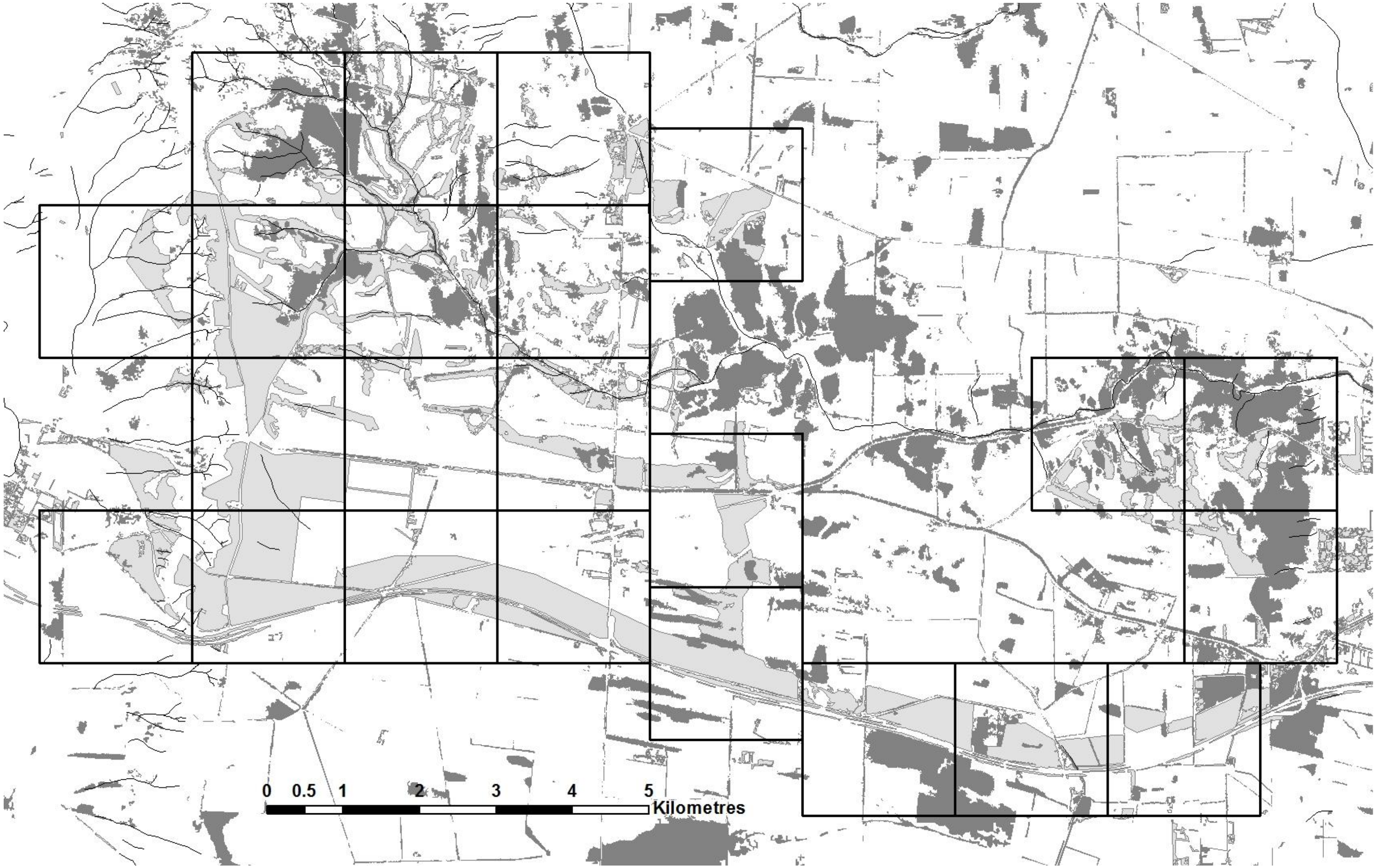


Fig. 2.1. Position of the 22 x 4 km^2 cells used for analysis in relation to revegetation (light grey), remnant vegetation (dark grey) and drainage lines (thin black lines). NB. The third cell down in the third column from the left was not included.

2.3.3 Bird surveys

In each 1 km^2 cell, all individual birds and species were recorded in systematic area searches of the revegetation. Birds were identified through calls or sight within ca. 50 m either side of the observer, until all the revegetation was covered. To ensure cells with different areas of revegetation were comparable, the revegetation was traversed at a consistent rate and birds detected in areas already sampled were not included. This meant that every part of the revegetation was sampled with equivalent effort and any differences due to area were real and not a result of spending more time in cells with more revegetation. All the revegetation in some cells could not be sampled due to restrictions on property access and these cells were either excluded (as mentioned above) or dealt with statistically to avoid underestimating species richness or abundance (see 2.3.6).

Searches were repeated at two different times of the year: Spring/Summer (October – December 2006) and Autumn/Winter (May – July 2007), in order to account for any variation due to seasonal migrants. Periods of strong winds (> 25 km/h) and high temperatures ($\geq 30^0$C) were avoided, and searches were undertaken from dawn to late morning or early afternoon depending on the conditions and associated bird activity. Each cell search lasted between 1.5 – 3.5 hours depending on the amount of revegetation present.

2.3.4 Response variables

For each 4 km^2 cell, woodland species richness and the relative abundance of 19 declining species were calculated by pooling the data from the corresponding 1 km^2 survey cells. Woodland species richness (herein Woodland Species) was calculated from only those species that occur more often in woodland than other habitats (i.e. open country or wetland birds were excluded; see Appendix 1 for classification), where woodland in this region was considered to be habitat containing trees and lacking shrubs characteristic of the other main treed habitat in the region - sandy mallee heath. This classification fits with that used in most other studies of woodland birds (Fraser *et al.* 2015), and therefore should enable reliable comparison of the results. In addition, also omitted were any woodland species that could not be consistently detected via the survey method (i.e. nocturnal species roosting in hollows or dense foliage; Owlet-nightjar, Southern Boobook and Tawny Frogmouth), and any exotic woodland species (e.g. European Blackbird, Spotted Dove, House Sparrow). These groups of species also correspond to the main groups excluded by many equivalent studies on woodland birds (Fraser *et al.* 2015). The declining species (Appendix 2) were regarded as those identified in Paton *et al.* (2004), and also the Red-capped Robin, which has not been classified as declining in the Mount Lofty Ranges but has been in other parts of southern Australia (e.g. Reid 1999). For each species, numbers recorded from both surveys were summed to give a measure of relative abundance and intensity of use (Martin & McIntyre 2007). The 19 species were only those present in ≥ 7 cells, as adequate statistical fit could not be obtained for species found in fewer than seven cells.

2.3.5 Landscape variables

Twelve landscape variables were used to explain the variation in the response variables and assess the research questions (Table 2.1). These were selected from an initial set of 23, the majority of which displayed high levels of inter-correlation (Appendix 3). The 12 variables selected were those that reduced the correlation as much as possible ($r_{Pearson} < 0.8$) while still answered the research questions. All were derived using ArcGIS 10.1 (ESRI 2012) and based on layers of revegetation and remnant vegetation that were manually digitised from 0.5 m resolution aerial photos taken in 2003 (Department of Environment & Heritage, South Australia).

Table 2.1. Descriptions and ranges for the landscape variables calculated for each 4 km^2 cell

Variable	*Description*	*Range*
Revegetation	Area of revegetation (ha)	34.4 - 185.3
Aggregation	% Revegetation area in largest effective patch (collective of patches with gaps ≤ 30 m)	33.7 - 100.0
Patch Size	Area-weighted average revegetation patch size for cell (ha)	4.1 - 57.0
Patch Shape	Area-weighted average shape index for revegetation patches, where 1 = circular and >>1 = elongated	1.5 - 4.0
All Remnant	Area of all remnant vegetation (ha)	3.1 - 139.6
Woodland Remnant	Area of woodland remnant (ha), or area of open woodland remnant (ha) depending on response	2.4 - 128.9 (2.4 - 80.1)
Drainage Length	Drainage length associated with revegetation (within 10 m of revegetation patches) (km)	0.0 - 4.5
Grazed	% Grazed revegetation, or area of grazed revegetation (ha) (dependent on response)	0.0 - 74.0 (0.0 - 52.0)
Habitat Diversity	Areal diversity of revegetation & remnant vegetation types, represented by the Shannon-Weiner index	0.2 - 1.7
Proximity Allocasuarina	Area-weighted average proximity to *Allocasuarina* remnant, as represented by the proximity index	0.0 - 583.2
Proximity Callitris	Area-weighted average proximity to *Callitris* remnant, as represented by the proximity index	0.0 - 255.3
Proximity OEW	Area-weighted average proximity to Open Eucalypt Woodland remnant, as represented by the proximity index	0.2 - 1042.5

Of these 12 variables, there were eight that were assessed for all response variables: Revegetation, Aggregation, Patch Size, Patch Shape, All Remnant, Woodland Remnant, Drainage Length, and Grazed. Revegetation and Aggregation were the variables used to determine whether only the total area is important or if plantings need to be close together. Revegetation represented the total area of the plantings and to avoid extraneous variables, this included all revegetation not just the 1970s plantings. These other plantings however, were small (all < 5 ha) or of similar age and structure, and therefore did not detract from the overall uniformity of the microhabitat. Aggregation was the measure of how close (or aggregated) patches of revegetation were, and rather than using the distance between patches (e.g. Average Nearest Neighbour) which does not incorporate their area, this was calculated as the percent area of revegetation in the largest effective patch – similar to the large patch index (e.g. Radford *et al.* 2005). An effective patch was a collective of patches with gaps of no more than 30 m, and was designed to incorporate functional connectivity, as individuals of many species had been observed regularly crossing between patches on either side of roads or railway lines - most of which were around 30 m wide. These patches also included remnant vegetation – for example, if two patches of revegetation were 100 m apart, but were connected by a patch of remnant, then these were considered to be part of the same effective patch. Therefore, to maintain relevance for each individual species assessed, remnant was adjusted for each to only those types considered to be usable habitat (e.g. patches of Heath were not included for the tree trunk and branch foraging Varied Sittella; see Appendix 2).

The average size and shape of revegetation patches (Patch Size and Patch Shape), were designed to address the questions of what size and shape plantings should be. Patch Shape was calculated using the shape index - $\mathrm{Perimeter}/2\sqrt{\pi \times \mathrm{Area}}$, where values of 1 correspond to more circular shaped patches and larger values to more elongated patches (Selwood *et al.* 2009, Mac Nally *et al.* 2010). This was used instead of an area to perimeter ratio, as it was relatively independent of patch size and therefore helped to reduce inter-correlation. Both of these measures were area-weighted to reflect the average of the majority of the area and maintain a landscape perspective. The total patch size and shape including remnant vegetation were also considered important, but total shape was highly correlated with remnant vegetation ($r_{\mathrm{Pearson}} > 0.75$), and therefore to avoid potentially reducing the effect of the remnant variables, these measures were not included.

The total area of all remnant vegetation (All Remnant) was used to examine whether plantings need to be associated with existing habitat. In addition, the area of existing woodland (Woodland Remnant) was used to determine if any kind of remnant could be beneficial to woodland birds or just woodland. Both variables were classified from vegetation types defined by observations made during the bird surveys, which were then sorted into categories based on a combination of structure and floristics (for details see Appendix 4). Woodland Remnant was comprised of Gum, Box, *Allocasuarina* and *Callitris* Woodland, and also Open mallee as many typical woodland species were also observed using this, including all the declining species assessed here. For some species, woodland remnant was refined further as Open Woodland Remnant (Appendix 2), according to observational experience of these species which suggested that denser woodland with high shrub cover was unlikely to be used. This was done to ensure that the Woodland Remnant variable reflected relevant habitat and to thereby give the best possible chance of finding any effect. Initially, the distance to all of these remnant types was to also be tested by including either the amount of adjoining remnant along with the area, or through proximity indices to represent both area and distance. However, there was very high correlation between All Remnant and Woodland Remnant for both these measures ($r_{Pearson} > 0.9$) and also with their corresponding area variables ($r_{Pearson} > 0.7$; Appendix 3), and therefore only remnant area was used to represent both the extent and distance to remnant vegetation. Finally, to represent the chance that only certain amounts of remnant may be needed, quadratic terms for each of these variables were also tested.

The length of drainage lines associated with the revegetation (Drainage length) was the variable used to ascertain whether planting in productive areas is valuable. This was defined as all the drainage length within 10 m of any revegetation patches and was based on 0.5 m resolution topographic layers (Geoscience Australia, 2003). Other variables related to productivity were also considered, including soils, topography and rainfall. However, these displayed very little variation across the plantings in the case of topography (all on flat or slightly undulating ground) and rainfall (< 50 mm gradient), or the variation in mapped layers did not correspond to that observed in the field in the case of soils. Hence, these variables were not included and only Drainage length was used to represent productivity.

The final main variable - the grazing status of the revegetation (Grazed), was not a landscape variable as such, but was included to complement the other landscape variables and account for additional variation in woodland birds. This was deemed important as areas of the 1970s plantings that have been subjected to grazing by stock generally have fewer plants, more dead trees, and a ground layer lacking in grasses and chenopods (J. Allan personal observation). As a result, these areas were perceived to be a different system that could provide habitat for a different set of species. Three species were thought to be dependent on areas of grazed revegetation (Jacky Winter, Red-rumped Parrot and Southern Whiteface), and hence for these species the grazing status was calculated as the area of grazed revegetation to represent the hypothesis that they needed a certain amount of this habitat. For all other species, grazing status was calculated as a percentage of the total area of revegetation in order to reflect either a negative influence or a positive but non-dependent influence. As grazed revegetation was expected to be a negative influence for some species but positive for others, it was included as a quadratic for assessing woodland species.

In addition to the eight main variables, one of four variables: Habitat Diversity, Proximity Allocasuarina, Proximity Callitris and Proximity Open Eucalypt Woodland (OEW), were also included for some of the response variables. Habitat Diversity was included for the woodland species response to establish whether planting in association with a variety of habitats would promote more species. The diversity was calculated using the Shannon-Weiner index ($-\Sigma p_i \ln p_i$, where p_i is the proportion of area in the *i*th habitat type) and was a function of all the remnant types defined in Appendix 4, and also of revegetation, which was regarded as being comprised of two different habitat types based on its grazing status (as mentioned above). As this variable was designed to test the hypothesis that more habitat types lead to more species, it was only included for the woodland species response and not for individual species.

The three proximity variables were included for several individual species to represent more specific remnant types, as it was envisaged they might respond more to these than to the general remnant variables. These were able to be represented by proximity indices, as unlike the general remnant measures they were not highly correlated with any of the other variables used ($r_{Pearson} < 0.6$). Proximity was calculated for each patch of revegetation, as the area of each patch of remnant divided by the square of the distance to that remnant and then summed across all revegetation patches (McGarigal *et al.* 2012). These figures were converted to area-weighted averages to reflect how much of the revegetation was proximal to each remnant type. For the Diamond Firetail, the proximity to remnant *Allocasuarina* woodland (Proximity Allocasuarina) was included, as this species is known to feed on *Allocasuarina* seeds (Ankor 2005). Proximity to remnant *Callitris* woodland (Proximity Callitris) was used for the Red-capped Robin and Yellow Thornbill, as many individuals of these species had been observed using patches of remnant *Callitris,* and even though they both use other vegetation types it was hypothesised this might lead to a higher relative abundance. Finally, the proximity to open eucalypt woodland (Proximity OEW) was used to represent the proximity of revegetation patches to hollows, as hollows were present in similar amounts in all patches of this vegetation type. This was included for the Brown Treecreeper and Southern Whiteface - two species known to use hollows for breeding (Higgins *et al.* 2001, Higgins & Peter 2002). Proximity OEW however, was not included for the one other hollow using species in the analysis (the Red-rumped Parrot; Higgins 1999), as this species was believed to be responding to hollows in scattered paddock trees outside patches of open eucalypt woodland, which could not be measured. As with the general remnant variables, quadratic terms for all of these proximity variables were also included to represent potential non-linear relationships.

2.3.6 Analyses

To assess the influence of the landscape variables on the bird responses, generalised linear models (GLMs) were constructed for each response and model averaging performed to determine the relative importance of each landscape variable on each response. Model averaging was used instead of the traditional approach of selecting a single best model, as this allowed the relative importance of each variable to be determined in the context of interactions with other variables in other models. The averaging process involves evaluating the weight of evidence for each model based on an information criterion and then determining the relative importance of each variable according to the weight of evidence of all the models in which it occurs (Burnham & Anderson 2002). Here, models were assessed based on the small sample corrected version of Akaike's Information Criterion (AIC_c) or the quasi equivalent ($QAIC_c$) where the response was overdispersed - see below (Burnham & Anderson 2002). The weight of evidence was represented by the Akaike weights (w_i) – the relative likelihood of the model in the set of models, which were summed (Σw_i) for each landscape variable to provide their relative importance (Burnham & Anderson 2002), and these values were then used to rank each variable for each response. As there were no groupings of variables that were considered more or less likely, averaging was conducted over all possible models with some support from the data (AIC_i - $AIC_{min} < 7$; Burnham & Anderson 2002). This was performed using the 'dredge' and 'model.avg' functions from the 'MuMIn' package (Barton 2014) in R (R Core Team 2014). Finally, averaged regression coefficients (ARC) - also provided by model averaging, were used to determine the average effect size and direction (+ or -) of each variable on each response.

As there was high multi-collinearity between the landscape variables, a second technique – hierarchical partitioning was used to determine the independent contribution of each landscape variable on each response. Hierarchical partitioning separates the joint (or collinear) effects from the independent effect of each variable by comparing the improvement in the goodness-of-fit in models with a given variable to those without (Mac Nally 1996, 2000). The 'partition' function in the 'hier.part' package for R (Walsh & Mac Nally 2013) was used to calculate the independent contribution (% I) for each landscape variable, with log-likelihood used as the goodness-of-fit measure. % I represents the independent contribution as a percentage of the total explained variance and this was used to rank the landscape variables in order of importance for each response. These ranks were then compared to those derived from model averaging to discern those variables that have both high weight of evidence and independent influence on the response.

For both these methods, GLMs were fitted with Gaussian errors for Woodland Species, while quasi-poisson errors were used for individual species as these followed a Poisson distribution but all showed evidence of overdispersion (dispersion parameter > 1). In the Woodland Species GLMs, landscape variables that exhibited logarithmic relationships with the response (Revegetation, Aggregation, and Patch Size) were log transformed to conform to the assumption of linearity of the explanatory variables for the Gaussian models. The fit of all GLMs was checked using diagnostic plots of residuals vs fitted values, normal Q-Q, scale-location, and residuals vs leverage provided by the 'plot' command for the 'glm' function in R. The total explained variation (explained deviance) for each global model was also calculated as a measure of model fit, to evaluate the effectiveness of the included variables at representing the response.

In addition, the GLMs also included an offset in order to account for the lower amounts of revegetation surveyed in some cells and the subsequent expected underestimation of the response. Specifically, the percentage of the total revegetation area surveyed - averaged across both survey periods was used as the offset. This was not included for the Woodland Species response though, as preliminary analyses with this variable as a covariate showed that it was having little effect (ARC = 0.07 ± 0.1) and when included as an offset had a large negative impact on the model fit. For the individual species GLMs, the offset was log transformed to correspond with the log link used for the quasi-poisson models.

Lastly, because the analysis units of the study (4 km^2 cells) were spatially clustered, tests for potential spatial autocorrelation were conducted on the models for each response. Tests were performed using the Moran's I statistic, which was calculated using functions in the 'spdep' package in R (Bivand 2014), following the process outlined by Dormann *et al.* (2007). Two neighbourhoods of 3000 m and 6500 m were tested, corresponding with immediate cell neighbours and two layers of cell neighbours. However, no significant effect of spatial autocorrelation on either scale was found for any of the response variables ($p > 0.05$), and therefore spatial autocorrelation was not considered to be a confounding factor in the analysis.

2.4 Results

There were 81 bird species recorded using the revegetation, and of these 57 were classified as woodland species (Appendix 1). The numbers of woodland bird species ranged from 27 to 44 per cell, while ranges for the relative abundances of individual species varied differently according to species (Appendix 2). Models for all responses displayed reasonable fit with the data, explaining 30-90% of the variation (Appendix 5).

2.4.1 Relative importance of landscape variables

Overall the rankings according to the summed Akaike weights (Σw_i) from model averaging, and those based on the independent contribution (% I) from hierarchical partitioning, were in broad agreement (Table 2.2). Drainage length had the most influence across all datasets being ranked in the top three variables for 12 out of the 20 responses, and also had consistently the highest Σw_i and %I values of all the landscape variables (Fig. 2.2g). For woodland species, the influence was slightly less than for those individual species where it was a top variable, with lower Σw_i (0.52) and a rank of four according to the independent contribution.

The second highest influence was from Revegetation, which was ranked in the top three for nine of the 20 response variables, and again had consistently high Σw_i and %I values, although generally less than Drainage length (Fig. 2.2a). Revegetation contributed around a third of the explained variation for Woodland Species, Brown-headed Honeyeater, Restless Flycatcher, Varied Sittella, White-browed Babbler and White-winged Chough.

Patch shape was the next best performing variable across the response variables, having relatively large Σw_i and %I and being ranked in the top two variables for four species (Fig. 2.2d). However, for two of these species (Yellow-rumped Thornbill and Yellow Thornbill), hierarchical partitioning suggested it was relatively unimportant (%I < 12). In contrast, the independent contribution was high for Dusky Woodswallow accounting for nearly 50% of the explained variation. For all of these species, Patch shape displayed negative regression coefficients indicating a trend toward more circular rather than elongated patches.

Three species (Brown-headed Honeyeater, Jacky Winter and White-browed Babbler) showed an influence of Patch Size with high Σw_i values (Fig. 2.2c). For the latter two species however, independent contribution was low, explaining < 12% for Jacky Winter and < 7% for White-browed Babbler. Conversely, Patch Size contributed over a third of the explained variation for Brown-headed Honeyeater and was the number one ranked variable for this species according to both Σw_i and %I.

Remnant variables overall displayed little influence, with only two species (Brown Treecreeper and Southern Whiteface) showing high positive effects; both to Woodland Remnant rather than All Remnant (Figs. 2.2e & f). In fact, All Remnant showed a relatively high negative influence for both these species and had negative averaged regression coefficients for most other response variables. The only other positive effect of remnant vegetation was from Proximity to Allocasuarina, which displayed moderately high Σw_i (0.47) and %I (24.1) for the Diamond Firetail (Fig. 2.2i). The inclusion of quadratic terms for any of the remnant variables did not improve their importance.

Only two species (White-winged Chough and Yellow-rumped Thornbill) showed a response to Aggregation, which was negative for the former and positive for the latter (Fig. 2.2b). For both though, the independent contribution suggested it was likely to be unimportant (< 11%).

Grazed was in the top two landscape variables and showed a moderately positive influence for three species (Jacky Winter, Red-rumped Parrot and Willie Wagtail; Fig. 2.2h). For the former two species Grazed represented the area of grazed revegetation.

None of the landscape variables had high Σw_i for the Hooded Robin or Silvereye ($\leq$ 0.4), inferring that none of the variables were overly important compared with the other responses. While some of the %I were moderately high for these species (> 20%), none were clearly high, overall suggesting that none of the landscape variables assessed were relevant in determining their relative abundance.

Table 2.2. Rankings of the landscape variables according to their summed Akaike weights (Σw_i) from model averaging and independent contribution (% I) from hierarchical partitioning for each of the 20 response variables. Only shown are the rankings for those landscape variables that had comparatively high relative importance (Σw_i > 0.5) and independent contribution (% I > 20). Diversity/Proximity rankings shown are for Proximity to Allocasuarina.

	Revegetation		Aggregation		Patch Size		Patch Shape		All Remnant		Woodland Remnant		Drainage Length		Grazed		Diversity/ Proximity	
	Σw_i	%I	Σw_i	%I	Σw_i	%I	Σw_i	%I	Σw_i	%I	Σw_i	%I	Σw_i	%I	Σw_i	%I	Σw_i	%I
Woodland Species	1	1											2	4				
Brown Treecreeper									1	1	2	2						
Brown-headed Honeyeater	2	2			1	1												
Common Bronzewing	2	2											1	1				
Diamond Firetail	2	3											1	1			3	2
Dusky Woodswallow							1	1										
Hooded Robin																		
Jacky Winter					3	3							1	2	2	1		
Red-capped Robin													1	1				
Restless Flycatcher	1	1					2	2										
Red-rumped Parrot															1	1		
Rufous Whistler													1	1				
Silvereye																		
Southern Whiteface									1	3	1	1						
Varied Sittella	2	1											1	2				
White-browed Babbler	2	2			3	5							1	1				
Willie Wagtail													1	1	2	2		
White-winged Chough	1	1	2	5					4	3			3	2				
Yellow-rumped Thornbill	5	2	2	5			1	4					1	1				
Yellow Thornbill	3	2					2	4					1	1				

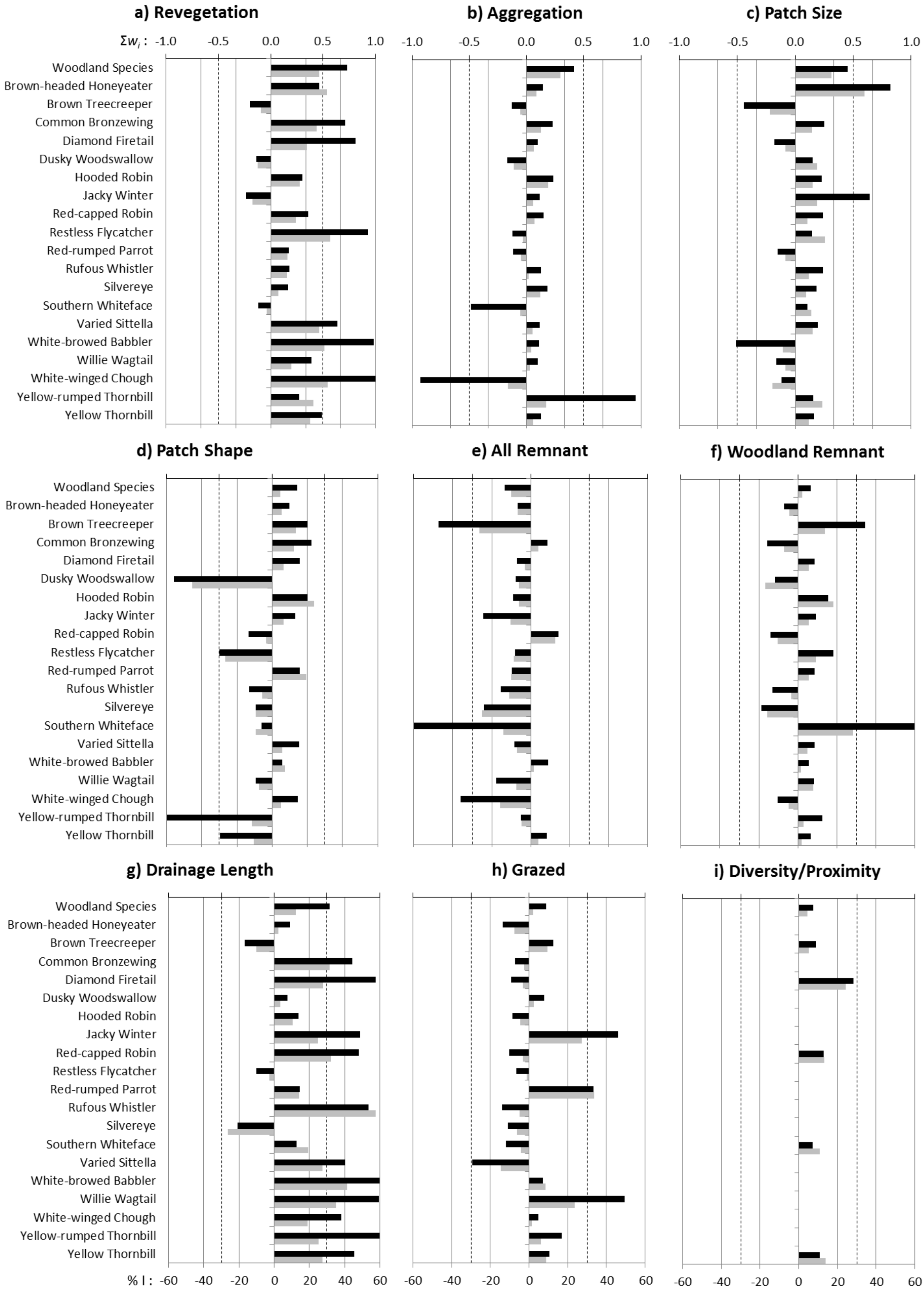


Fig. 2.2. Relative importance of the landscape variables for each of the 20 response variables according to the summed Akaike weights (Σw_i – black bars, top scale and dashed vertical lines) and independent contribution (%I – grey bars, bottom scale and vertical lines). The last landscape variables where included, represent: Diversity for Woodland Species, Proximity OEW for Brown Treecreeper and Southern Whiteface, Proximity Allocasuarina for Diamond Firetail, and Proximity Callitris for Red-capped Robin and Yellow Thornbill. Values for landscape variables with -ve averaged regression coefficients were inverted to indicate their negative effect.

2.5 Discussion

2.5.1 Important landscape features

Drainage length was clearly the most influential landscape feature across the responses, being the most important feature for the majority of the individual declining species and also influencing the total number of woodland species. This result fits with studies of birds in remnant vegetation in similar areas along gullies and riparian zones that have also found that these areas have greater numbers of species and higher abundances of individual species (e.g. Mac Nally *et al.* 2000b, Woinarski *et al.* 2000, Palmer & Bennett 2006). As suggested by these studies, the increased biodiversity associated with drainage lines is likely to be due to higher moisture levels and deeper, richer alluvial soils, leading to increased plant growth and subsequently higher levels of resources (e.g. nectar and invertebrates). Moreover, these effects have also been found in revegetation with two studies showing that plantings in these areas have higher numbers of species (Lindenmayer *et al.* 2010, Munro *et al.* 2011), and the results here reinforce these findings. Importantly though, the results in this study also suggest that planting in association with drainage lines not only can increase the total number of species but increase the abundance of declining species. This indicates that reconstructing habitat along drainage lines will be important in reversing the declines in species of conservation concern, and further emphasises the importance of these areas as valuable places to reconstruct habitat.

The area of revegetation was the other standout feature and overall was more important than aggregation, and also the size and shape of the plantings. This conforms to the majority of previous research that indicates the extent of habitat is more important than configuration (Bennett *et al.* 2006), and suggests that only the total area of reconstructed habitat is important. However, most of revegetation in this system is highly contiguous (e.g. many cells had around 100% of habitat aggregated within 30 m) and also part of large block shaped patches, and hence the ranges in these features were not large. Furthermore, the total amount of habitat across the study area (both revegetation and remnant) is about 25% which is close to the 30% threshold at which configuration effects have been found to diminish (Andren 1994, Fahrig 1997, Radford *et al.* 2005). Therefore, while this result reinforces the importance of establishing large amounts of reconstructed habitat, it may not be an adequate assessment of the effects of configuration (particularly at lower levels of habitat), and it should be treated with caution.

Total area of revegetation however, was not more important than size and shape for two species (Brown-headed Honeyeater and Dusky Woodswallow), while shape was the second most important feature behind revegetation area for the Restless Flycatcher, potentially indicating species specific responses. For the Brown-headed Honeyeater, patch size was the most important feature which given their large area requirements (Chapter 3) makes sense as when habitat is distributed more closely in large patches movement is likely to be more efficient leading to lower energy requirements (*sensu* Hinsley 2000) and therefore potentially a higher number of birds. The effect of shape for the other two species though, was less clear. Similar to the Brown-headed Honeyeater, the Restless Flycatcher is a large area user (Chapter 3) and therefore an association with more circular patches of habitat on the surface seems logical for the same energy efficiency reason outlined above. However, as with the Dusky Woodswallow, the Restless Flycatcher was observed using thin strips of vegetation and scattered trees in paddocks away from large circular patches (unpublished data). This effect may therefore be at least partly spurious, possibly caused by the low numbers of elongated patches in the plantings at Monarto (as mentioned above). Nonetheless, there still may be an effect of higher abundances in more circular shaped patches and therefore as a matter of precaution if habitat is to be reconstructed for these species this effect should be taken into account.

Of the other variables, Grazed had the most influence, indicating the importance of including relevant microhabitat features in landscape level analysis. As stated, this variable was not included to assess the effect of grazing *per se,* but the different microhabitat features in these areas, and as expected it benefited some species with the Jacky Winter, Red-rumped Parrot, and Willie Wagtail showing positive responses. All these species are characteristic of more open woodland typical of the grazed revegetation. For the Red-rumped Parrot though, the response may not be directly to grazing but an association of grazed areas to water points, as being a granivore they require regular access to water (Higgins & Davies 1996). A variable representing water points was sought prior to analysis for this reason, but was unable to be obtained. Including this variable in future analyses however, may help to provide a greater understanding of the landscape requirements for this species and other granivores (e.g. Diamond Firetail).

In addition to these three species, the Southern Whiteface was also expected to show a positive response to grazed revegetation, but this was unimportant in their analysis. This was surprising, as all of the records of this species were in revegetation that had been grazed or within about 50 m of grazed areas (unpublished data). Southern Whiteface did show a response to woodland remnant though (see below), and it may be that the combination with grazed revegetation was not represented adequately. There were several cells for instance, with large amounts of grazed revegetation but very little woodland remnant where whitefaces were not recorded. An interaction between the two variables was tried in preliminary analyses, but did not change the result, and this may be because there were other cells where there were no whitefaces with large amounts of woodland remnant and grazed revegetation, albeit separated by large distances. Future analyses with this species might therefore consider including variables indicating the proximity of woodland remnant to grazed revegetation as well as all revegetation.

The remaining landscape features were the remnant variables, which had little effect overall. Such a finding may seem surprising as there is much research showing there are nearly always more birds in remnant than revegetated areas due to a lack of key microhabitat features in revegetation (e.g. leaf litter, bark, forbs; Loyn *et al.* 2007, Barrett *et al.* 2008, Munro *et al.* 2011), and therefore it would be expected that revegetation associated with remnant vegetation would have more birds. Indeed, some studies have also found increased species in revegetation associated with remnant vegetation (Kavanagh *et al.* 2007, Lindenmayer *et al.* 2010). But, unlike other revegetated areas, the plantings at Monarto are used by the vast majority of woodland bird species in the region (Paton *et al.* 2010a), many of which reside in home ranges almost completely comprised of revegetation (Chapter 3). These factors suggest the microhabitat features are of equivalent quality to remnant vegetation, and may therefore explain why remnant has little effect in this system. This is an important finding, as it infers that if the microhabitat is sufficient then for most species reconstructed habitat does not need to be placed near remnant vegetation.

On the other hand, the Brown Treecreeper and Southern Whiteface did show a response to woodland remnant. This was not surprising though, as both these species require hollows – a microhabitat feature that is currently missing in the revegetation in this system. It was a little unexpected therefore that proximity to Open Eucalypt Woodland was not important for these species, as this was the vegetation that contained the hollows. But, both species also use other woodland remnant types for foraging (e.g. *Allocasuarina* and *Callitris*; Barker 2007, Hoffmann 2011) and therefore a combined effect of all woodland may have been responsible. The negative effect of all remnant for both these species was also unexpected, but was probably caused by the majority of remnant vegetation having relatively high shrub cover, which is likely to impede both these species as they spend much time foraging on the ground (Higgins *et al.* 2001, Higgins & Peter 2002). Similarly, this may also explain the negative response shown to all remnant for other ground-foraging species like the Jacky Winter and White-winged Chough.

In addition to these two species however, there were two more declining species (Chestnut-rumped Thornbill and Painted Button-quail) which may also require woodland remnant. These species were not included in the analysis as they were both only recorded in the revegetation in four cells and subsequently adequate model fit could not be achieved. But, both were always found in cells with large patches of remnant vegetation, and have only ever been recorded in the revegetation in close proximity to remnant vegetation (Richards 2008; Allan & Paton unpublished). This is logical for the Chestnut-rumped Thornbill as it is also a hollow user (Higgins & Peter 2002), while Painted Button-quail are usually found in habitat with a dense layer of branch and leaf debris (Marchant & Higgins 1993), features which are probably unlikely to be common in 30 year old woodland revegetation. Therefore, if habitat is to be reconstructed for these two species then it may need to be placed in association with remnant vegetation, at least as a precautionary measure until their landscape requirements can be properly assessed.

The only other positive effect of remnant vegetation was Proximity to Allocasuarina for the Diamond Firetail. Such a finding was also not surprising, because as mentioned previously this species is known to feed on *Allocasuarina verticillata* seed (Ankor 2005). However, this may not be an effect of remnant vegetation as such, as Diamond Firetails also frequently use patches of *Allocasuarina* in the revegetation (unpublished data), which was a feature unaccounted for in the analysis. The effect may have instead come about because the patches of planted *Allocasuarina* are smaller than those in remnant (J. Allan personal observation) and this potentially led to more seeds and more birds around remnant patches. Nonetheless, this reinforces the importance of *Allocasuarina* for this species, a fact that should be taken into account when reconstructing their habitat, whether it is through planting near existing patches or including this plant in revegetation.

2.5.2 Practical implications

Overall the results presented in this study suggest that to maximise biodiversity outcomes for most woodland birds reconstructed habitat should be placed in large block-shaped areas associated with more productive land around drainage lines, while for some species these areas also need to be near existing woodland. In contrast however, most current revegetation is placed in small, isolated areas on unproductive land. For example, Harris (1999) investigated revegetation in the Tungkillo area of the Mount Lofty Ranges and found that despite numerous revegetation activities, most of the plantings were very small with nearly half < 1 ha, and were planted as thin strips, typically situated along ridges or on poor soils, and were generally isolated from remnant vegetation and other plantings. Unfortunately this practice appears to be widespread with similar patterns also found in other regions (e.g. Wilson *et al.* 1995, Smith 2008), and this clearly needs to change if biodiversity outcomes are to be maximised from the limited funding available. Such a change will be difficult though, as areas associated with drainages are likely to be productive not only for biodiversity but for agriculture, and therefore obtaining large amounts is likely to be costly, and this may bias revegetation aimed at being cost-effective away from these areas. However, the results presented here suggest that these areas will provide the most benefit, and this needs to be factored into cost-benefit models (e.g. Westphal *et al.* 2007, Crossman &

Bryan 2009, Lethbridge *et al.* 2010, Wilson *et al.* 2011) to weigh the increased biodiversity against the increased cost, and allow the most cost-effective places in which to reconstruct habitat to be truly determined.

2.5.3 Future improvements

While this approach has successfully identified important landscape features, there are some potential drawbacks that could be improved in future. First, many relevant microhabitat features were not included, and this may have meant a large portion of the variation was missed, thereby hampering the assessment of landscape features. For instance, the influence of grazed revegetation and proximity to *Allocasuarina* illustrated the importance of microhabitat level features for some species, and a lack of relevant microhabitat features could explain why none of the landscape features were influential for the Hooded Robin or Silvereye. Both of these species are likely to respond to microhabitat. For example, the abundance of Silvereyes is known to fluctuate with the availability of nectar and fruit (Higgins *et al.* 2006), while observational studies suggest Hooded Robins require plants that provide lateral branches from which they can pounce, and also a range of ground substrates (e.g. Gillespie 2005, Antos & Bennett 2006, Northeast 2007). Variability in these features independent of landscape features may have been driving some of the variation for these two species, and hence hindered the assessment of important landscape features. However, there is no simple method of including microhabitat features, as the traditional approach of using small samples will suffer from the same potential problems as the bird sampling and will not necessarily capture all the variation, while sampling over entire landscape units would be infeasible. It may be possible though to use high resolution aerial photographs which are now available for the region to rapidly discern relevant microhabitat features over broad areas. For example, large shrubs and trees can be differentiated, along with several important individual plant species (e.g. *Callitris gracilis, Allocasuarina verticillata*, *Casuarina glauca*; J. Allan personal observation), and these could be used to generate relevant structural and floristic variables. Including these variables may improve the variance explained for some species and in turn enhance the assessment of landscape features.

Another potential drawback of this approach is the lack of temporal replication, as surveys were only conducted twice and whilst these were over two seasons any differences between years may not have been captured. Short term datasets for instance can be unrepresentative over longer time frames (Maron *et al.* 2005), and therefore the data here should be interpreted with some caution. There is some evidence though, that the data here is broadly representative of longer timeframes. For example, the number of woodland species and abundances of declining species in one of the 2 x 2 km cells is similar to an overlapping site (RV4) that was surveyed regularly (27 times) over an eight year period (Chapter 4 and Paton unpublished). Forty woodland species were recorded in this study while 49 were recorded in the 27 surveys, and the relative abundances of most declining species were broadly similar given differences in area (Appendix 6). Of the species that were not detected in this study, six out of nine could be considered nomadic or seasonal migrants to the region, and the three sedentary species were recorded in < 30% of the long term surveys potentially indicating a low preference for the area (Appendix 7). Therefore, the data appear to represent most of the resident species found in the area, and thus any negative effects associated with the single year of sampling are unlikely to be large. Nonetheless, there were differences, particularly with the migrants and species less common to the area, and hence any future applications of this approach should increase replication in order to account for these and help improve the assessment of landscape features for these species.

Finally, although important landscape features have been identified, another potential drawback to the approach used in this study is the lack of predictive models, i.e. exactly how much drainage length is required, and what size patches? Answers to these questions will be necessary in order to assess the range of options in areas of landscapes with these features. The reason predictive models were not developed here was the influence of potential boundary effects. For instance, while variables were checked to be representative of the real attributes, the values for the variables would have been capped (e.g. average size of patches in a cell may have been 50 ha but a portion of the patch areas may have also been outside the cell), and therefore any predictions would have been inaccurate. Fixing this problem would involve selecting more ‘closed’ landscapes in which most of the features are wholly contained. However, this was very difficult for Monarto due to the high level of habitat contiguity, and the arrangement and scale of cells were already adjusted as much as possible without increasing the scale further and reducing the sample size to an unacceptable level. Hence, developing predictive models may need to be done in other regions where relatively closed landscapes can be found. Given that a number of important features have been identified though, an alternative solution could be to specifically target these features at finer scales in order to discern the thresholds at which they become important. Birds at varying distances from drainage lines for example, could be examined and used to determine how close habitat needs to be to these areas. Even though thresholds would be developed at a patch rather than landscape scale, this information could be included in reconstruction models with the knowledge that these features are in fact important at landscape scales. Such information could then also be included in spatial planning models (e.g. Thomson *et al.* 2007, Westphal *et al.* 2007, Thomson *et al.* 2009) to allow the formulation of plans for multiple species that may have differing requirements (Westphal & Possingham 2003).

2.5.4 Conclusion

This study has determined a number of landscape features that are important for woodland birds, and in particular for a range of declining woodland birds. Given these results were obtained by landscape scale sampling there can be increased confidence that they reflect a high degree of the variation in patchily distributed species. Moreover, as many of these results reinforce those obtained using small samples over broader scales, there can be confidence that they also reflect broader patterns, and on the other hand, that previous results also correspond to the requirements of more patchily distributed species. Hence, these results can be used to prioritise where and how future reconstructed habitat is placed with the knowledge that they do in all likelihood represent the requirements of the majority of the woodland bird community, and therefore that biodiversity outcomes will indeed be maximised from the funding available.

Chapter 3
Minimum area requirements for reconstructed habitat patches based on home ranges

3.1 Abstract

The reconstruction of habitat is viewed as the mechanism that will curb biodiversity declines caused by habitat loss, but the stakes are very high as many species are likely to become extinct if suitable habitat is not created. Hence, there is a need to ensure that every piece of reconstructed habitat is sufficient to meet species requirements, and a central part of this will be determining how much habitat groups of animals need at local scales, in a patch or set of patches. A number of patch size guidelines for reconstructed habitat already exist, but are based on assessing the use of individual patches and determining the minimum sized *patch* that a species will occupy, rather than the area used by groups of animals *per se*. These represent useful guidelines for many species and are easily replicated over broad scales, but may not reveal the entirety of the area required by groups of some highly mobile species able to use multiple patches. Instead, in this study, home ranges were used to estimate the entire area used and determine the minimum patch areas needed to provide sufficient habitat for groups of woodland birds in the Mount Lofty Ranges, South Australia. Home ranges from three woodland bird species anticipated to be large area users were obtained using radio-telemetry in existing reconstructed woodland, and these were combined with home range data from a range of other woodland species studied previously in the same system. Eighty-five home ranges from a total of 16 species were obtained, and the area of habitat used within home ranges ranged from 166 ha to just under 10 ha. These areas are much larger than nearly all existing revegetation efforts, and therefore should serve as a wakeup call as to just how much reconstructed habitat may be required by groups of some birds. This knowledge can contribute to ensuring that every piece of habitat created in future is sufficient, and thereby give the best possible chance of securing the persistence of these species.

3.2 Introduction

Habitat reconstruction has been hailed as the solution to reversing biodiversity losses associated with habitat clearance (Saunders & Hobbs 1995, Vesk & Mac Nally 2006). The stakes are extremely high though, as many species are facing imminent extinctions and their survival in many regions depends entirely on the reconstruction of appropriate habitat. For example, in southern Australia despite the cessation of habitat clearance, woodland birds are continuing to decline markedly (Ford *et al.* 2009, Ford 2011), with a number of species already having disappeared from several regions (Ford *et al.* 2001) and many more predicted if suitable habitat is not created (Recher 1999). Therefore, there is a need to ensure every piece of habitat created is sufficient to fulfil species requirements, as there are unlikely to be any second chances.

One of the fundamental elements in ensuring habitat reconstruction is sufficient will be providing a suitable amount of habitat. Much attention has been given to the amount of habitat required at broad scales, with population models and metapopulation research producing estimates for the area needed in large patches and landscapes to maintain viable populations (e.g. Hanski 1994, Bulman *et al.* 2007, McCoy & Mushinsky 2007). Similarly, fragmentation studies have provided estimates of habitat required over landscapes to prevent the loss of entire species (e.g. 20-30%; Andren 1994, Fahrig 1997, Radford *et al.* 2005), which have been used to set landscape scale restoration targets by regional authorities (e.g. 30% of ecosystems to be restored; AMLR NRM Board 2014). However, less attention has been given to the area of habitat required within landscapes at local scales to support individuals and groups. Such a perspective will be vital, as it is a fundamental principle of ecology that individuals and groups – which are the base units of species and populations, need a certain amount of habitat to meet their energy requirements, and give them the ability to survive and reproduce (*sensu* Schoener 1968). Indeed, in the field of spatial conservation planning, estimates of areas used by individuals and groups of several species have been used as the basis for examining the capacity of landscapes to support viable populations (e.g. Goldingay & Possingham 1995, Nicholson *et al.* 2006, Nicholson & Possingham 2007). Determining the minimum area required by individuals and groups at local scales, will therefore be essential to ensure reconstructed areas provide a sufficient amount of habitat, and guarantee species are able to survive and persist at both local and broader scales.

Several patch size estimates have been produced to indicate the minimum area of reconstructed habitat required to support species at local scales within landscapes (Lambeck 1999, Freudenberger 2001, Major *et al.* 2001, Watson *et al.* 2001, Brooker 2002, Mac Nally & Horrocks 2002, Huggett *et al.* 2004). However, all of the estimates for reconstructed habitat at this scale have been based on determining the minimum sized *patch* that a species will occupy, rather than the precise area used by a *group* of animals *per se*. These estimates are extremely important in that they reflect the minimum patch area required to overcome negative effects of fragmentation (e.g. edge effects), and the area required by many species that are only able to use individual patches and unable to easily cross gaps in habitat. But, they may not necessarily reflect the full extent of the area required by individuals or groups of some species with higher mobility, which are able to use multiple patches on a daily basis. For instance, individuals of species with high mobility may perceive fragments of habitat in a landscape not as discrete islands, but as a series of fine-grained patches, and therefore use a range of patches to satisfy their area requirements (Kotliar & Wiens 1990, Andren 1994, Wiens 1995). Birds in particular are highly mobile, and indeed, individuals of many species have been shown to use more patches as their habitat becomes increasingly fragmented and the size of individual habitat patches decreases (see examples in Andren 1994). Assuming groups of such species only used single patches in these cases would not account for the area of habitat used in other patches, and only attribute the area required to the smallest sized patch in which they were found. Therefore, using single patches to estimate the area requirements for groups of some more highly mobile species may not reveal the entirety of the habitat required.

An alternative method for assessing the minimum area required by groups of animals is to measure the area of habitat within their home range. A home range represents the area traversed by an animal in its normal daily activities (Burt 1943), and as such is not necessarily tied to any single patch and simply represents the area that the animal uses. Hence, this method may reveal more of the total area required by individuals and groups of species able to use multiple patches. For instance, home ranges have been used to discern amounts of remnant habitat required to be protected for some highly mobile species of birds in fragmented forest (e.g. Wiktander *et al.* 2001, Bilney *et al.* 2011). Tracking groups of animals and documenting all the area used however, will be more intensive than simply recording the presence of species in individual patches, and therefore may reduce the ability to replicate effectively over broad areas. This method though, may be a way of complementing existing well replicated estimates, and ensuring any resulting areas of reconstructed habitat are sufficient to meet the needs of both highly mobile and less mobile species.

In this study, home ranges were used to determine the area of habitat required by a range of woodland bird species in order to guide habitat reconstruction in the Mount Lofty Ranges region of South Australia. Many woodland bird species in the region have suffered severe declines associated with widespread habitat loss, and already 8-10 species have disappeared (Paton *et al.* 2004, Szabo *et al.* 2011). Existing research has suggested that a range of typical and declining woodland birds in the region are highly mobile and likely to use large areas spread across multiple patches of habitat (Paton *et al.* 2004, Willoughby 2005). Hence, understanding the full extent of areas required by groups of these species, and ensuring any reconstructed areas established are sufficient to meet their requirements, will be crucial in reversing their declines.

The main aim was to determine the area used by species with some of the largest area requirements, so these could be used as the estimate for the minimum area required by the majority of the woodland bird community (a form of the focal species approach; *sensu* Lambeck 1997). The second aim was to obtain as many area estimates from as many woodland bird species as possible, particularly declining species, in order to validate the largest area estimate, and also so these individual species estimates could be used to guide any species-specific restoration programs targeted at these birds.

3.3 Methods

3.3.1 Study area

The area requirements of woodland birds were examined at Monarto, about 60 km east of Adelaide between the eastern flank of the Mount Lofty Ranges and the Murray River (35°3'S, 139°2'E - 35°9'S, 139°13'E). Monarto was chosen for this study as it contains around 1850 ha of reconstructed woodland planted in the mid to late 1970s as part of preparations for a satellite city to Adelaide that was eventually abandoned (Paton *et al.* 2010b). This system was important for determining area requirements of woodland birds, firstly as it contains many of the species that have declined in the rest of the region (Paton *et al.* 2004). Second, because reconstructed areas can take many decades to match the resource levels of remnant vegetation (Vesk *et al.* 2008), birds may require more area to survive in younger re-established habitat than in fully mature remnant vegetation. Therefore, understanding how much of this 30 year old piece of ‘developing’ habitat is required, may be additionally beneficial to ensure woodland birds are able to use reconstructed areas before they reach full maturity.

3.3.2 Study species

To assess the area required by the majority of the woodland bird community, three species were selected: Restless Flycatcher (*Myiagra inquieta*), Varied Sittella (*Daphoenositta chrysoptera*) and Brown-headed Honeyeater (*Melithreptus brevirostris*). These species were anticipated to be some of the larger area users in the woodland bird community based on previous observations and experience of these birds. All three species are also considered to be declining in the Mount Lofty Ranges (Paton *et al.* 2004), but present in reasonable numbers in the reconstructed woodland at Monarto (Chapter 2). The other species chosen were those that had been studied previously at Monarto. These comprised a number of typical woodland species and also declining woodland birds (see below).

3.3.3 Data collection

Data on the three anticipated larger area users was collected using radio-telemetry. As birds are known to restrict their area use in the breeding season (see Schoener 1968), tracking was undertaken during the non-breeding season in April - August 2008, in order to obtain the maximum area used. Birds were captured with mist nets at a number of locations throughout the revegetation. At each location, all birds captured from the three species were banded with unique combinations of colour bands, and one bird of each species was fitted with a radio transmitter weighing ≤ 3% of its body weight (0.3 g for the Brown-headed Honeyeater and Varied Sittella, and 0.6 g for the Restless Flycatcher; Advanced Telemetry Systems, Isanti, Minnesota). The transmitters were attached to the interscapular area of the back with superglue and a small piece of gauze after trimming an area of feathers the size of the transmitter to 1-2 mm of the skin. Each bird was subsequently tracked using an Icom R10 RX5 receiver (Bio-Telemetry Tracking, Australia) fitted with a yagi antenna (Sirtrack, New Zealand) for at least seven days over a period of 2-4 weeks, depending on the transmitter battery life. Locations of birds (fixes) were recorded with a handheld Global Positioning System (GPS) when each bird with the transmitter or another member of its group was sighted (determined by the colour bands). In instances where a bird could not be sighted (usually due to movement onto inaccessible land), locations were extrapolated via triangulation (< 1% of records). Initially, fixes were to be obtained at intervals of 1-2 hours, but early tracking revealed birds made short < 1 hour visits to areas on the fringe of their range, and therefore birds were tracked as continuously as possible from dawn till dusk to avoid missing any areas of the home range.

For the other species previously studied at Monarto, data were compiled and collated from a number of projects in order to extract estimates of area requirements. These data also came from the reconstructed woodland, but also included data from neighbouring remnant vegetation. Some of the data were obtained by radio telemetry as with the three species tracked in this study, but most was collected by observing birds with unique combinations of colour bands. The majority of the locations were collected in continuous bouts similar to this study, but were also complimented with additional opportunistic data, either sightings or captures in mist nets. Most data came from targeted studies conducted over a couple of seasons in one year, but combined with the opportunistic data some spanned several years. Where possible, groups were identified for each species based on information provided in the corresponding studies or by observing overlapping patterns in space and time in the data. To ensure the data were reflective of the area requirements of each species, groups or individual birds with a minimum of 20 records collected over at least three days were used to generate home ranges. The only exception was a Golden Whistler, for which only 15 records were obtained. The data for this bird was included because it was collected over a long period (i.e. on nine separate days over three years) and also because very little area usage data exist for the species (Higgins & Peter 2002).

3.3.4 Home range estimation

Home ranges were estimated using Minimum Convex Polygons (MCPs). Although MCPs have been criticised due to their inability to distinguish and exclude unused areas in home ranges (Worton 1987, Harris *et al.* 1990, Powell 2000), there were a number of reasons for their use here. First, because the majority of the data for many of the home ranges was collected over short timeframes, delineating unused areas would have been inappropriate as it would be impossible to determine if an area within a home range was not used at all, or merely not used within the tracking period. Also, as the data used to derive some of the home ranges were almost completely opportunistic and therefore not intensive, it did not provide an adequate understanding of the interior use of these home ranges, and hence other methods that delineate unused areas would have likely resulted in underestimates. Therefore, given the aim was to determine the area required to ensure sufficient habitat, it was thought better to err on the side of overestimating area and ensuring species are catered for, rather than going for a more accurate estimate but potentially underestimating area requirements. Ideally, data would have been collected both intensively and over longer timeframes and more advanced methods used, but as with any similar study this was unrealistic due to short transmitter life, and time and budgetary constraints. In any case, excluding non-vegetated areas (i.e. non-habitat; see 3.3.5), the majority of the area in each home range was used, particularly for the three species tracked in this study, and the unused areas within home ranges were relatively small (e.g. Appendix 8).

To construct MCPs for each home range, all of the locations obtained for each group of birds were used (100% MCPs). The convention is usually to remove locations on the edge of home ranges by including only 95% of the area used, as these are deemed to be "occasional sallies" and areas not used in normal activities (Powell 2000). However, this practice can result in locations that are actually biologically important to an animal being excluded (Powell & Mitchell 2012), and therefore, once again because the aim here was to avoid underestimates, all locations were included as a matter of precaution. In this case, nearly all extremities of the home ranges of the three species tracked in this study were used for foraging anyway, demonstrating their importance and need for inclusion.

3.3.5 Habitat area

The area of habitat within each 100% MCP was calculated by summing the area of all the woodland contained within the home range using a combination of the intersect and spatial join tools in ArcGIS 9.3 (ESRI 2008). The woodland areas were based on vegetation layers manually digitised from 0.5 m resolution aerial photography taken in 2003 (Department of Environment & Heritage, South Australia) and classified into remnant or revegetation according to observations on the ground during the data collection. The habitat area estimates for each species were interpreted as the maximum area of habitat recorded in a home range for that species. This contrasts with the approach taken by patch based studies of using the minimum area (e.g. Lambeck 1999, Watson *et al.* 2001) or low probabilities of occupancy (e.g. Brooker 2002, Huggett *et al.* 2004), and also home range studies that cite averages as the area required to support individuals or pairs of a species (e.g. Wiktander *et al.* 2001, Glenn *et al.* 2004). However, given many of the species in question are declining and may face extinction if suitable habitat is not reconstructed, using the maximum area was deemed necessary to ensure the best possible chance of providing sufficient habitat and securing their persistence into the future. The average and minimum habitat areas contained in a home range were however also provided for comparison.

3.4 Results

Habitat area estimates were derived for 85 home ranges from 16 woodland bird species (Appendix 9). Eight home ranges were obtained for the three target species in this study. These consisted of four for the Brown-headed Honeyeater, three for the Varied Sittella and only one for the Restless Flycatcher, due to difficulty in catching these birds. The other 77 home ranges were extracted from other studies and comprised estimates for 13 species, including seven listed as declining in the Mount Lofty Ranges.

The largest area of habitat recorded in a home range was 165.8 ha for a group of Varied Sittellas, while the lowest maximum area was 8.2 ha from a group of Superb Fairy-wrens (Table 3.1). Habitat area estimates for all species varied considerably with at least several fold differences between the minimum and maximum estimates. All areas estimated were comprised mainly of revegetation except for a few home ranges documented in remnant vegetation away from the revegetated areas (Appendix 9).

Table 3.1. Habitat area estimates for 16 woodland bird species ordered by the maximum area of habitat documented for each. The habitat area corresponds to the amount of woodland vegetation contained in each minimum convex polygon home range.

Species	Home Ranges	Habitat Area (ha)			Authors
		Avg	Min	Max	
Varied Sittella*	3	122.6	67.9	165.8	Current study
Restless Flycatcher*	1	156.8	156.8	156.8	Current study
Rufous Whistler*	5	41.9	19.7	90.6	Hunt (2011) & Paton (unpublished)
Brown-headed Honeyeater*	4	52.8	26.4	77.0	Current study
Owlet-nightjar	11	17.4	2.1	66.1	Hlava (2005) & Crossfield (unpublished)
White-browed Babbler*	11	15.0	4.1	38.1	Paton (unpublished)
Hooded Robin*	7	19.4	8.7	36.6	Gillespie (2005), Northeast (2007) & Paton (unpublished)
Yellow-rumped Thornbill*	9	12.3	1.7	35.2	Davill (2001) & Paton (unpublished)
Diamond Firetail*	5	11.3	2.0	22.9	Ankor (2005) & Paton (unpublished)
Golden Whistler	1	18.4	18.4	18.4	Paton (unpublished)
Southern Whiteface*	6	10.6	6.2	16.9	Hoffmann (2011)
Chestnut-rumped Thornbill*	4	12.9	8.7	16.0	Richards (2008) & Paton (unpublished)
Brown Treecreeper*	3	9.4	7.4	11.5	Barker (2007) & Paton (unpublished)
Variegated Fairy-wren	1	9.5	9.5	9.5	Pethybridge (2003) & Paton (unpublished)
Red-capped Robin	11	5.1	1.1	9.5	Gillespie (2005), Northeast (2007) & Paton (unpublished)
Superb Fairy-wren	3	6.2	4.8	8.2	Pethybridge (2003) & Paton (unpublished)

* Denotes declining species according to Paton *et al.* (2004)

3.5 Discussion

3.5.1 Minimum patch area requirements

The largest amount of habitat contained in a home range for any of the species was 166 ha, which suggests this is the minimum area required to support at least one group of all the woodland birds included here. This estimate is similar to the 200 ha recommended by Major *et al.* (2001) as the area required to increase the chance of occupancy for a number of bird species of conservation concern in the wheat belt of New South Wales. It is larger however, than the 80 ha cited by Mac Nally & Horrocks (2002) and the 100 ha from Watson *et al.* (2001), and is much larger than the 10-31 ha prescribed by the majority of other patch-based studies (Lambeck 1999, Freudenberger 2001, Brooker 2002, Huggett *et al.* 2004). All of these studies included all of the largest area users here, potentially illustrating differences between the methods used. However, these inconsistencies could also be due to differences in habitat quality between the remnant vegetation in which previous studies were conducted and the revegetation used here, or general differences in habitat quality between regions. Nonetheless, the result here reinforces the existing findings that 10s to 100s of hectares of habitat will be required to support a range of woodland birds. Moreover, as the estimate here represents the area required by just one group of some species (i.e. Varied Sittella and Restless Flycatcher), it not only suggests this is the area needed to increase the chance of occupancy, but suggests this is the minimum area required to have any chance of these species occupying an area.

As this estimate represents the area required by groups of a range of woodland birds, it can be used to guide the size of reconstructed areas for multi-species restoration programs where the aim is to provide habitat for as many species or declining species as possible. However, it is important to recognise that this result was derived from a limited number of birds, over short timeframes, and therefore should be treated as the absolute minimum area required, as it still could be an underestimate. In particular, the one Restless Flycatcher home range obtained contained only 10 ha less habitat than the maximum area recorded and given there were no other estimates to compare with this, it is entirely possible that this could be a small estimate for the species and an underestimate for the largest area used across all the species. In addition, there is the potential that there are other species with larger area requirements than those considered here. Birds that rely on larger, vertebrate prey and birds that have larger body mass generally have larger territories (Schoener 1968). Woodland birds such as raptors, owls and frogmouths fall into these categories and may have much larger area requirements than those documented here. In the absence of estimates for these species though, the estimates presented here should represent a reasonable guide to the minimum patch size requirements of a large portion of the woodland bird community.

In terms of the other, smaller area estimates presented here, as with the largest estimate, nearly all of these are greater than previous patch-based estimates derived for the same species (all except the 100 ha estimate for the Hooded Robin in Watson *et al.* 2001). As mentioned, these differences could be due to differences in the methods or a range of other factors that are impossible to separate. But, as also stated, these estimates represent the area required by one group and hence are a guide to the minimum areas likely to be required to facilitate their presence in a patch or local area. This is important, as many of these species are of conservation concern, not only in the Mount Lofty Ranges but in other regions across southern Australia (Olsen *et al.* 2005). Hence, these estimates could be used to set the minimum patch area required for restoration programs specifically targeted at any of these species to ensure they are able to provide for at least one group and contribute to their overall recovery. In addition, the range of different species estimates presented can also help managers discern what is practical in a given management scenario. For instance, landholders or local authorities may not have the capacity to provide habitat for a larger area user in a particular region (e.g. Watson *et al.* 2001), but instead can use the list of area requirements to find a species they are capable of providing for. In this way, these estimates can also ensure that outcomes are optimised from the resources and opportunities available.

There are however, a number of important clarifications on all of these estimates in relation to their potential use. First, even though there was a large degree of variability in most species estimates and the minimum and average areas were much smaller than the maximum, using these instead is strongly discouraged. This is because variation in home range size is commonly caused by differences in habitat quality between home ranges, with higher quality habitat producing smaller home ranges and vice versa (e.g.

Conner *et al.* 1986, Pasinelli 2000, Anich *et al.* 2010). Therefore, an adequate understanding of the factors that contribute to habitat quality would be needed to create sufficient habitat in a smaller area. However, while some important habitat features have been identified for these species (Marchant & Higgins 1993, Higgins *et al.* 2001, Higgins & Peter 2002, Higgins *et al.* 2006), a quantitative understanding of exactly what constitutes quality habitat, and hence the ability to recreate it, has not been established yet. Therefore, this reinforces the need to use the maximum habitat area documented for each species, in order to guarantee the best possible chance of providing sufficient habitat, at least until the drivers behind habitat quality have been found and quality habitat can be created with confidence.

Second, although home range overlap was unable to be measured (tracked groups were separated by several kilometres), it is likely that many of the home ranges documented were exclusively used by the corresponding groups. For instance, all groups of all three species tracked in this study exhibited signs of territorial defence (fights, chasing and incessant calling around the borders of home ranges), and incursions by individuals that were not part of the tracked groups were rare (identified by being unbanded or a banded bird from another group; unpublished data). Also, many of the species included from other studies were observed as being territorial or are generally known to be (e.g. Rufous Whistler, Hooded Robin, Brown Treecreeper; Higgins *et al.* 2001, Higgins & Peter 2002). Hence, the areas occupied by these birds should be regarded as exclusively used territories, and it cannot be assumed that the areas cited will provide for any more than one group.

Finally, while many of these estimates are based on a number of groups, they were only collected in one region. This may be important, because as mentioned habitat quality is a major determinant of home range size, and as the range of all of these species spans multiple rainfall gradients and different vegetation types (Marchant & Higgins 1993, Higgins *et al.* 2001, Higgins & Peter 2002, Higgins *et al.* 2006), there are likely to be differences in quality and hence area requirements between regions where they occur. Estimates of the habitat area required based on home ranges from other regions where habitat reconstruction is proposed or at least in those where habitat is quite different, may therefore be needed in order to safeguard against potential underestimates. Until then, these estimates can at least serve as a starting point to give the best possible chance of providing sufficient amounts of habitat.

3.5.2 Implications for practice

Two overall practical implications arise from these results: 1) in order to support groups from a range of woodland birds - including highly mobile, large area users, 100s of hectares of habitat will need to be created, and 2) ensuring the presence of groups of some lower area using species may also require around 10 ha of habitat. However, the vast majority of current reconstructed areas fall well below 100 ha and many also fall below 10 ha. For example, the range of planting sizes reported in studies examining faunal use of revegetation is usually around 1 – 20 ha (e.g. Cunningham *et al.* 2007, Lindenmayer *et al.* 2010, Munro *et al.* 2011), many of which are isolated from other plantings and remnant vegetation (Mac Nally *et al.* 2010). Based on the data here, areas at the upper end of this range would only cater for species in the lower half of the estimates presented, and are about 8 times or 140 ha less than the maximum estimate of 166 ha. Moreover, areas at the bottom of this range would not even ensure sufficient habitat for species with the lowest habitat area estimates below10 ha. Clearly, if reconstructed areas are to provide sufficient habitat, they will need to be significantly larger than existing efforts. Securing and establishing such large areas may be difficult though, due to reluctance of landholders to give up equivalent portions of productive land (e.g. Watson *et al.* 2001) or alternatively due to the probable high cost of purchasing large tracts of land, and also due to the expensive nature of revegetation (e.g. Schirmer & Field 2002). One possible solution to reduce the area required would be to focus on incorporating existing patches of remnant vegetation into reconstructed areas as much as possible, rather than establishing areas comprised mainly of new habitat. Such a strategy would not only save on the area required but also have the added benefits of aiding colonisation (Shanahan *et al.* 2011a) and providing slow developing resources (e.g. hollows, fallen timber; Vesk *et al.* 2008) to adjacent plantings. Whatever the mechanism, there needs to be a much more strategic approach in order to achieve the areas estimated here.

3.5.3 Other considerations

How should the area be distributed?

The habitat area estimates presented here are a vital step towards providing sufficient habitat, but there are other considerations in relation to how that area should be distributed before they can be implemented on the ground. First, should the area be distributed in single or multiple patches? All the birds from the three species tracked in this study used multiple patches (Appendix 9), suggesting this could be an option. This raises another question though: how far apart can the patches be? The patch based studies that have provided previous area estimates also provide estimates as to how far apart patches should be based on the most isolated occurrence for a species, and these are used to recommend the distances at which patches should be created (e.g. Watson *et al.* 2001, Brooker 2002). However, while these data can provide information on the maximum distance that a species is able to move (i.e. dispersal between meta-populations), it does not answer how far apart patches need to be for regular, day to day movements in a home range. Such a perspective will be important, as there is evidence that habitat patches need to be within a certain distance in home ranges to avoid decreases in foraging efficiency and reproductive success due to the increased energetic costs of movement (e.g. Hinsley 2000, Hinam & St. Clair 2008). Hence, inter-patch distance needs to be assessed using home ranges, i.e. measuring the gaps between patches used and establishing a threshold over multiple home ranges for a species. Unfortunately, this was not possible for this study due to the large contiguous plantings established in the 1970s, and even though the species tracked here used multiple patches, most of these were close together (usually only around 50 m apart). In order to assess this effectively, patch distances would need to be measured in home ranges from more fragmented areas with patches at varying distances, before multiple patches could be planted with confidence. Moreover, just because the species in this study can use multiple patches does not mean other species have this ability. The probability of movement of other common woodland birds for example, has been shown to decrease markedly when there are gaps in the vegetation (e.g. Robertson & Radford 2009), and therefore if habitat is to be established for species other than the largest area users, then multiple patches may not be an option. In addition, small patches have higher amounts of habitat edge which has been associated with increased levels of nest predation (e.g. Luck *et al.* 1999) and also greater numbers of Noisy Miners which aggressively exclude many bird species (e.g. Piper & Catterall 2003). Hence, given this and the aforementioned reasons, establishing multiple patches may not be a viable option and planting single large patches may be the safest approach.

Another consideration needed in relation to the distribution of the area is what shape should the reconstructed areas be - blocks or strips? Again, Monarto was not appropriate for assessing this aspect, as most of the revegetation was planted in large blocks, and very few of the home ranges documented contained thin strips. Logically though, thin strips will suffer from similar negative edge effects to small patches, and there is also evidence that they can be sub-optimal habitat (e.g. Major *et al.* 1999), which points to the need to plant in large blocks. However, this does not reveal the amount of habitat edge able to be tolerated in home ranges, and if particular shapes and dimensions are required to constitute a viable home range. This information could be gained by examining the amount of edge and core habitat in home ranges and establishing thresholds for species. Until then, a precautionary approach to establish large blocks and avoid creating any kind of narrow strip of vegetation should give the best possible chance of providing suitable habitat.

What constitutes 'habitat'?

The area estimates presented here were based on 'habitat' broadly defined as woodland, however in order for these estimates to be effective the makeup of the vegetation, i.e. microhabitat, will also need to be considered. Woodland birds are diverse in their microhabitat requirements and every species has a unique set of features that they use, which sometimes are contrasting between species. For example, ground pouncing insectivores such as robins require open areas with few shrubs in order to access the ground , while other birds like fairy-wrens need shrubs; also, granivorous species such as Diamond Firetails need grassy areas, but other ground foraging birds like White-winged Choughs require extensive litter layers (Antos *et al.* 2008). Therefore, care needs to be taken to ensure the area established contains habitat for the species concerned. For reconstructed habitat targeted at single species, this is a relatively simple task of ensuring the area contains features needed by that species, but for multi-species programs this is a more challenging exercise of satisfying the range of unique and sometimes contrasting requirements. The revegetation at Monarto is an example of an area where this has been successfully (albeit accidentally) achieved, with home ranges of the largest area users containing at least one home range of nearly all the other species considered here (unpublished data). Key drivers behind this success can be found in Chapter 4, and these along with any other information on microhabitat requirements can be used to ensure the areas estimated here are capable of sustaining groups of the corresponding species.

Improving accuracy to avoid potential overreach

The emphasis of this study has been to avoid underestimating area requirements to ensure the best possible chance of providing sufficient habitat, but in future there may also be a need to avoid overestimates. As previously mentioned, establishing large areas of habitat will be a challenging task as it will require significant funding and access to large areas of land, therefore avoiding overestimates as well as underestimates - i.e. improving accuracy, could also be important in making these areas more achievable. To improve accuracy, more advanced home range estimators like kernel density (Worton 1989) or local hull based methods (e.g. Getz & Wilmers 2004, Downs & Horner 2009) would need to be used in combination with an increase in the temporal span of sampling to definitively establish areas used and not used in a home range. As stated, this was not possible in this study due to short transmitter battery life and the intensive nature of the data collection. However, it is possible to reduce the pulse rate on transmitters to extend the battery life from ca. 12 days to 60 days for the species tracked in this study (Advanced Telemetry Systems, Isanti, Minnesota). Tracking intensively for all this time would be infeasible, but birds could be followed at regular intervals, e.g. every few days over the period to obtain amounts of data similar to those here whilst also gaining a greater temporal scope. For species easily recaptured, this could also be repeated in different seasons and years to give an even longer temporal span. Kernel or local hull based methods could then be used to delineate areas of use with confidence, and while the increased temporal span may also increase the estimates, this will prevent overestimates and ensure no unnecessary area is estimated.

Population level area requirements

While the estimates presented here represent the area required to provide sufficient habitat for the occupancy of groups of birds, on their own they will not be sufficient to secure the persistence of a species. For example, the broad scale area estimates mentioned in the introduction from the metapopulation models (Hanski 1994, Bulman *et al.* 2007, McCoy & Mushinsky 2007) and fragmentation research (Andren 1994, Fahrig 1997, Radford *et al.* 2005), indicate networks of sufficient habitat for groups at the landscape scale will be also be required to ensure the persistence of entire species and populations in the face of stochastic events such as fire, floods or drought. As the estimates here represent the area required by groups, which as mentioned are the base unit of species and populations, they could be used to calculate the area needed to support viable populations along with general rules on the number of groups required (Shanahan & Possingham 2009). Additional information on the geographic extent of the population of each species and the amount of extant habitat would also be needed, to determine how much suitable habitat already exists and the amount that needs to be reconstructed. Once this information is obtained though, these estimates could be used to complement existing broad estimates for the occupancy of species in landscapes (i.e. 20-30% habitat cover) and help ensure the survival of species at both local and broad scales.

3.5.4 Conclusion

The results presented here reveal just how much habitat groups of birds need, and should serve as a wakeup call as to the amounts of reconstructed habitat required to support groups of woodland birds. While this exposes the shortcomings of current efforts, this knowledge, in conjunction with information at landscape and microhabitat scales, will help ensure that every future piece of habitat created is sufficient to meet species requirements, and thereby give the best possible chance of securing the persistence of these species.

Chapter 4
Identifying key microhabitat features for habitat reconstruction using the fine-scale distribution of woodland birds

4.1 Abstract

Habitat reconstruction is required to reverse declines in biodiversity throughout the world, but the opportunities will be limited and it is vital suitable habitat is created in the chances provided. A crucial part of recreating suitable habitat will be understanding the microhabitat requirements of species, and a large number of studies have already successfully identified a range of key microhabitat features. However, nearly all of the research directed at identifying features required for habitat reconstruction has been based on microhabitat features documented in samples positioned without knowledge of the exact locations used or unused by species. This method has clearly been effective and is easily replicated over broad scales, but may not capture absolutely all the variation in habitat use, and therefore could potentially overlook some of the finer details of habitat requirements. Here, an alternative more focussed and detailed approach was used to determine the fine scale distribution of woodland birds and identify the key microhabitat features required for habitat reconstruction in the Mount Lofty Ranges region, South Australia. The fine scale distribution of species richness of all woodland birds and declining woodland birds were used to guide the sampling of microhabitat features across a range of richness values in patches of existing reconstructed habitat. Generalised linear mixed modelling and model averaging were used to determine the relative importance of 12 microhabitat features in driving the patterns. The diversity of overstorey plant species was the most important feature for both all woodland bird species and declining species, followed by the diversity of ground layers and the evenness of overstorey and understorey for all woodland birds, while lower plant densities were also important for declining species. These results suggest that reconstructed habitat should include a range of overstorey plant species, mixed with understorey plants, established at lower densities and incorporating a

variety of ground layers. As these features were identified based on the fine scale distribution of habitat use, there can be increased confidence that they reflect some of the finer variation in microhabitat requirements, and therefore, when combined with existing results incorporating broader variation, this can also give much greater confidence that any resulting reconstructed habitat will support the species concerned.

4.2 Introduction

Habitat loss is the primary factor causing species declines throughout the world (Vié *et al.* 2009), and the reconstruction of habitat is widely regarded as essential in order to counteract these declines (Saunders & Hobbs 1995, Recher 1999, Vesk & Mac Nally 2006). The opportunities for habitat reconstruction however, will be limited as revegetation is expensive (Schirmer & Field 2002) and obtaining the land required will be difficult due to the associated loss of agricultural production (Dorrough *et al.* 2008). Therefore, it is imperative that the most is made of the opportunities provided and suitable habitat is reconstructed.

A crucial part of recreating suitable habitat will be understanding the microhabitat requirements of species, that is, features within patches of spatially contiguous habitat or vegetation types (i.e. grassland, pine-oak woodland; Hutto 1985). Microhabitat features provide the resources needed for the activities of foraging, reproduction and predator avoidance that are vital to the survival and persistence of species. Indeed, many studies have shown that species can exhibit strong associations with particular microhabitat features, such as vegetation structure (MacArthur *et al.* 1962), floristics (Holmes & Robinson 1981), and specific microhabitat elements like fallen timber (Mac Nally *et al.* 2001) and hollows (Gibbons & Lindenmayer 2002). Identifying key microhabitat features will therefore be critical if species are to use reconstructed habitat.

A large number of studies have already successfully identified a range of microhabitat features specifically for the purpose of guiding future habitat reconstruction (Barrett & Davidson 1999, Twedt *et al.* 2002, Arnold 2003, Law & Chidel 2006, Kavanagh *et al.* 2007, Loyn *et al.* 2007, Maron 2007, Barrett *et al.* 2008, Selwood *et al.* 2009, Lindenmayer *et al.* 2010, Mac Nally *et al.* 2010, Gardali & Holmes 2011, Munro *et al.*

2011, Yen *et al.* 2011, Lindenmayer *et al.* 2012). However, all of this research has been based on assessing microhabitat features documented in samples positioned without knowledge of the exact locations used or unused by species, i.e. in plots or transects randomly placed in the broader patches, plots or transects used to record species use. This approach has clearly been effective, but may not capture absolutely all the variation, as animals can display distinct patterns of habitat use at very fine scales; finer than that of the areas used to record species. For instance, home range studies have shown that animals use the space they occupy disproportionately, and exhibit 'core areas' of concentrated use (Samuel *et al.* 1985, Harris *et al.* 1990, Powell 2000). It is possible that without knowledge of such variation, a microhabitat sample could be situated on the edge of a core and a non-core area, and therefore habitat use would vary throughout. All the features recorded in the sample would be attributed to the same level of habitat use even though they may be less or more frequently used, or not used at all. With no knowledge of the exact distribution underlying habitat use, it is impossible to separate these, and therefore, some of the finer details of microhabitat requirements could be overlooked.

Using fine scale variation in habitat use to guide the measurement of microhabitat features, on the other hand, may help capture some of the finer details of microhabitat. For example, studies on foraging behaviour of individuals have illustrated many species are extremely selective in their use of microhabitat features within the same habitat, with strong preferences for certain plant species (e.g. Holmes & Robinson 1981, Recher & Majer 1994) or substrates (e.g. Holmes *et al.* 1979, Recher 1989); and indeed, at least two studies have used such preferences to infer the microhabitat features required for habitat reconstruction (Gabbe *et al.* 2002, Shanahan *et al.* 2011b). Similarly, research on the use of habitat within home ranges and territories has identified a range of features associated with core areas of use, and importantly that the features in these areas are distinct from the rest of the home range/territory; in terms of features sampled across the whole area (e.g. Chamberlain & Leopold 2000, Barg *et al.* 2006, Broughton *et al.* 2014) or at random points (e.g. Anich *et al.* 2012), highlighting the extra details using fine scale variation can provide. Documenting the fine scale distribution required to distinguish this level of detail though, will undoubtedly require much greater levels of sampling compared to documenting habitat use at the scale of plots, transects or patches. Hence, this may reduce the ability to replicate widely over time and space.

However, any extra detail on microhabitat requirements will help create the most suitable habitat in the opportunities provided, and therefore using fine scale variation may be a valuable tool for identifying the key microhabitat features required for habitat reconstruction, and one that can complement existing findings determined at broader temporal and spatial scales.

Here, the fine scale distribution of woodland birds was used to identify key microhabitat features for habitat reconstruction in the Mount Lofty Ranges region, South Australia. In southern Australia, habitat loss has been particularly severe with many regions having lost over 90% of their original vegetation (Saunders *et al.* 1991, Ford *et al.* 2001). These losses have involved the disproportionate clearance of woodland systems, and associated with this has been a major decline in woodland birds (Ford *et al.* 2001). Moreover, despite the cessation of clearance, in many regions woodland birds continue to decline (Ford *et al.* 2009, Mac Nally *et al.* 2009, Ford 2011). The Mount Lofty Ranges are one such example, with 8-10 species already having disappeared and around 50 more continuing to decline (Paton *et al.* 2004, Szabo *et al.* 2011). Broad scale reconstruction of woodland habitats is urgently required in order to reverse these declines and prevent further extinctions (Paton *et al.* 2004, Paton 2010, Szabo *et al.* 2011). Identifying the complete range of species microhabitat requirements will be vital if suitable habitat is to be reconstructed and these species are to persist in the region.

Specifically, the aim was to determine the key microhabitat features required to support a range of woodland bird species. To do this, species richness was used as the measure of habitat use, and microhabitat features were assessed across a range of richness values to determine the features that facilitate more species. Two measures of richness were used: the richness of all woodland bird species, and the richness of declining woodland bird species, which was also used to determine if these birds had any specific requirements further to those for all species.

4.3 Methods

4.3.1 Study area

The study was undertaken in existing reconstructed woodland in the Monarto region, approximately 60 km east of Adelaide on the eastern edge of the Mount Lofty Ranges (35°3'S, 139°2'E - 35°9'S, 139°13'E). The reconstructed woodland was planted in the mid to late 1970s as part of plans to establish a satellite city to Adelaide in the area, and was designed for the purposes of dust and erosion control, and to make the area more aesthetically pleasing for human habitation (Paton *et al.* 2004). Over 1800 ha of cleared, agricultural land was revegetated with around 600 000 trees and large shrubs comprising approximately 250 species originating from all over Australia and particularly Western Australia, as well as several from overseas. The plants were established in a standard fashion as tubestock placed 4-6 m apart in rows also 4-6 m apart (Paton *et al.* 2004, Paton *et al.* 2010b). However, despite the standard manner of planting, there was considerable variation in both the floristic and structural mix established. For example, some areas were planted with a diverse array of tree and shrub species, compared to others comprised only of 1-2 species of either trees or shrubs, which in turn also led to differences in the current density due to species specific mortality (e.g. Fig. 4.1). As a result, the microhabitat features also vary considerably throughout the plantings, and this offered an ideal opportunity to determine which of this diverse range of features, are the key microhabitat features required by woodland birds.

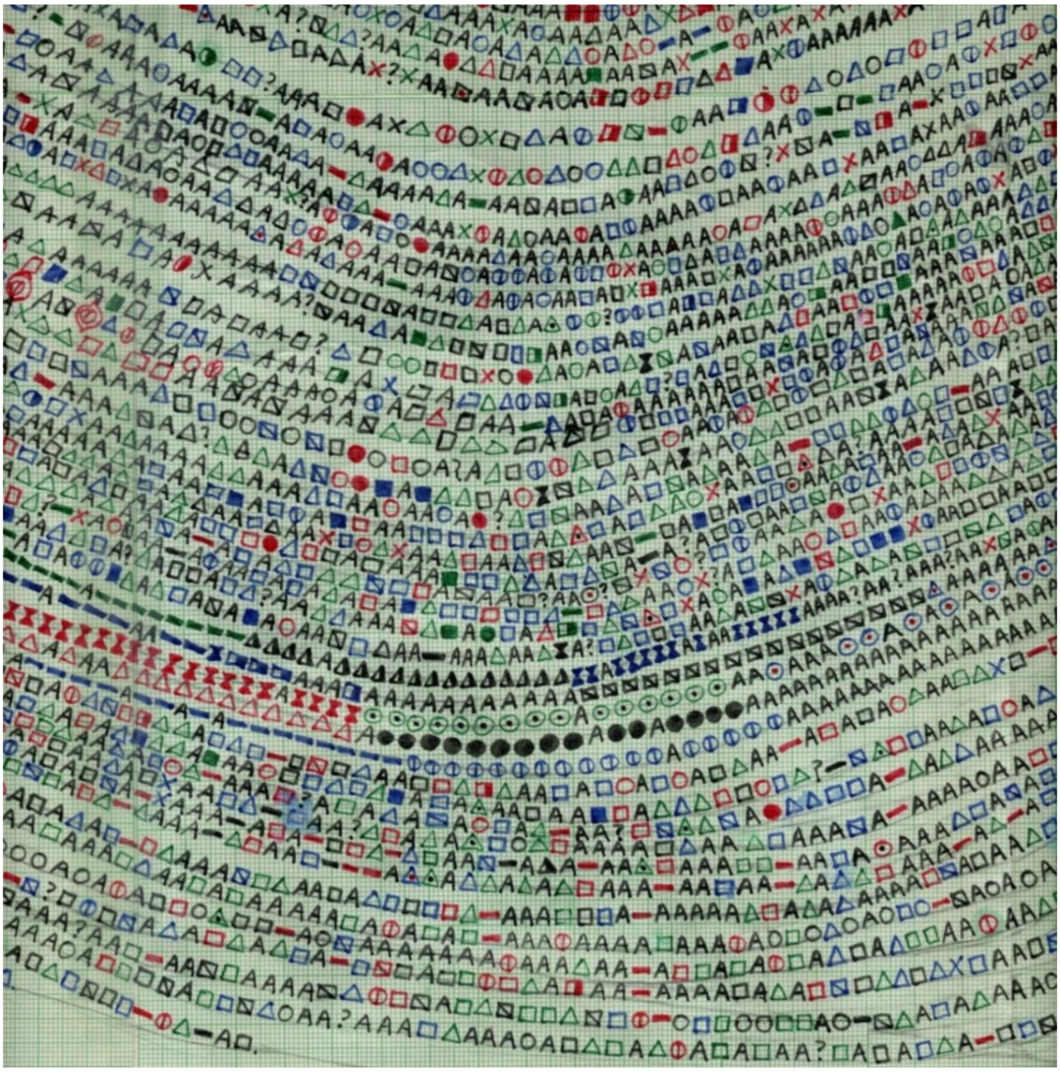


Fig. 4.1. Hand drawn map of the plants established in one section of the Monarto plantings, illustrating the variation in the floristic and structural mix throughout the plantings. Each symbol corresponds to a plant, with different types representing different species. 'A' reflects a dead plant. From unpublished data sourced from the former Woods and Forests Department, South Australia.

4.3.2 Survey sites

Five patches of revegetation were selected to undertake regular bird surveys (Fig. 4.2). These patches were large in size, ranging from 40 to 60 ha, in order to accommodate a range of different microhabitat features. The locations of the patches were also selected to maximise the variation in microhabitat features and use between site variations in planting, recruitment and chenopod regeneration. Two of the sites (RV1 and RVB) contained small portions of remnant vegetation that were also surveyed for birds, although for the purposes of this study only the revegetation was considered.

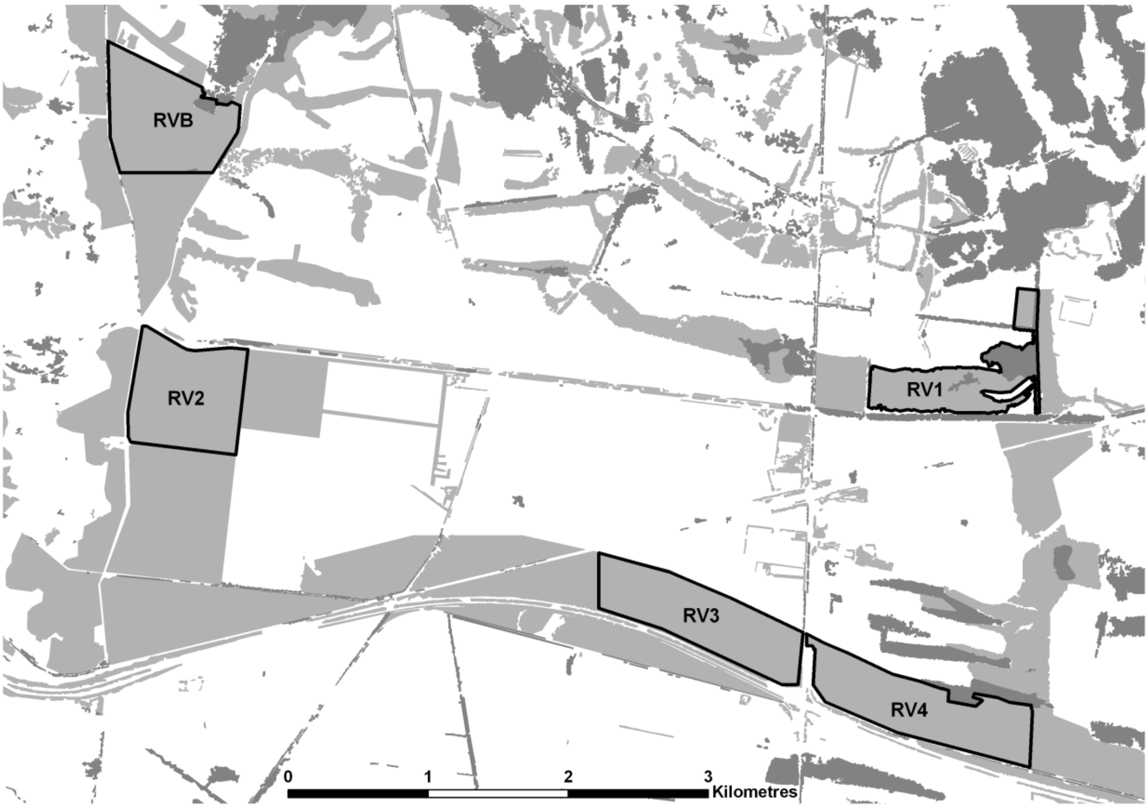


Fig. 4.2. Patches where regular bird surveys were conducted (◘). Revegetation is shown in light grey and remnant vegetation as darker grey. Site names are indicated in bold.

4.3.3 Bird surveys

The fine scale distribution of birds in each site was ascertained using 'systematic area search mapping' (Paton unpublished). This technique involves identifying the locations of birds with a Global Positioning System (GPS) in structured area searches of a defined patch. Locations are recorded for every individual bird or group of birds of the same species at the point of observation, such that each search produces a set of points or map, representing the locations of all birds within that patch. Patches are traversed systematically until the entire area is covered, with all areas only being searched once and birds detected in areas already surveyed not included in the sample. Speed is kept consistent to ensure all areas are sampled with equal effort. In this study, points were only recorded when GPS accuracy was estimated to be 5-10 m to ensure adequate representation of the area the birds were using. Also, the start location and search direction were alternated between surveys to ensure that any patterns were not an artificial construct of the search, and there was no bias associated with time of day.

In order to detect any spatial patterns, searches were conducted 13 times per site during 2007 (65 surveys in total). Surveys were spread over three seasons to account for fluctuations in species richness caused by seasonal migrants, and potential temporal differences in use of habitat by species. Five surveys were conducted in autumn and four each in winter and spring. Each survey commenced upon sunrise and usually lasted 4-6 hours, depending on the number of birds present. To ensure adequate detection, days of warm (predicted maximum above 25°C) or inclement weather were avoided. Sites were not surveyed during summer due to increased temperatures impacting on the ability to complete searches before detection decreased. All surveys were conducted by J. R. Allan, except for three of the 13 at site RV4, which were performed by D. C. Paton.

Points from surveys were restricted to woodland species, where woodland was considered to be habitat containing trees, and woodland birds defined as those species that are present more often in woodland than other habitat types (i.e. open country or wetland; see Appendix 10 for classification). This classification is consistent with the major factors used to categorise woodland birds (Fraser *et al.* 2015), and should therefore ensure that the results are comparable with most equivalent studies. In addition, nocturnal birds roosting in artificial hollows or nest boxes (e.g. Owlet-nightjar), or in dense foliage (e.g. Southern Boobook) were excluded due to concerns these species were unlikely to be consistently detected by the search method (i.e. every nest box or hollow was unable to be checked, and birds sitting quietly in dense foliage were likely to have been overlooked). The introduced House Sparrow and European Blackbird, are not of conservation concern and therefore were also removed. These groups of species also matched those excluded by other studies of woodland birds (Fraser *et al.* 2015). To reduce the effect of noise, records were also restricted to those species that were present in three or more surveys at each site, as these species could not be assumed to be resident in the survey sites and responding to local microhabitat features. Finally, declining species were defined according to Paton *et al.* (2004).

4.3.4 Distribution surfaces

To represent the distribution of birds, points from all surveys were pooled for each site and converted to density surfaces in ArcGIS 9.3 (ESRI 2008). Surfaces were generated using the point statistics tool and were comprised of 5 m pixels. Species richness values for each pixel were calculated from points within a radius of 50 m, which was the scale that best fitted the patterns observed in the field. For display purposes, each surface was smoothed to reduce visual artefacts of the search radius by averaging over a 50 m radius using neighbourhood statistics in the spatial analyst toolset. The surfaces were then displayed as stretched values using two standard deviations.

4.3.5 Microhabitat sample locations

Using the distribution surfaces for each site, a range of locations were selected to characterise the microhabitat features being used by woodland birds. The richness surfaces were first classified into five classes using the natural breaks Jenks classification in ArcGIS 9.3, to ensure samples were spread across the spectrum and to enhance the ability to detect any thresholds. Then, a large number of random points were generated at ≥ 50 m apart to accommodate the microhabitat survey area (see 4.3.6

below) using Hawth’s Analysis Tools (Beyer 2004). The three points in each class that varied the least across surveys and exhibited the lowest spatial variation in richness were chosen using the coefficient of determination and standard deviation respectively, in order to select areas that were temporally consistent in their use and had a relatively consistent level of richness throughout their area. This left a theoretical maximum of 15 locations for each response at each site (3 points x 5 classes) and 75 locations in total across all five sites. However, for some classes only two points were able to be selected due to insufficient area. The overall number of locations also varied depending on the degree of overlap between the samples selected for each variable. In total, 61 sample locations were selected for all woodland bird species and 60 were selected for declining woodland bird species.

4.3.6 Microhabitat surveys

At each sample location a range of microhabitat features were recorded using a ‘circular’ grid of 59 cells centred around planting positions in the revegetation, over a ca. 25 m radius (Fig. 4.3). This technique capitalised on the standard spacing between rows and plants within rows throughout the revegetation, and was designed to facilitate the timely collection of the data. Planting density varied slightly, and hence the size of grids also varied. However, any variation was minimal, and was partly offset by habitat variables being calculated per unit area (see 4.3.7). Initially, the intention was to survey habitat features at the 50 m scale at which the bird surfaces were generated, but logistical constraints forced the radius to be reduced.

For each planting position, a range of plant details were recorded for plants (if present), including the species, minimum and maximum canopy height, condition (canopy intact), and percent fallen. Around each planting position the makeup of ground layers was recorded in ‘equidistant’ cells, i.e. with boundaries defined as halfway between rows and halfway between plants within rows. The percentage covers of all ground layers were estimated visually, including bare ground, litter, fallen timber, grass, and small shrubs (< 1 m; predominantly chenopods).

In addition, the locations of all plants were recorded using a differential GPS (dGPS) to give an idea of the clustering of plants and variation in horizontal structure (see 4.3.7 below). The dGPS provided sub-metre accuracy and overcame the problem of plants in the revegetation being spaced around 5m apart combined with a traditional non-differential GPS accuracy of 5-10 m. Plant locations were linked to the corresponding plant details, enabling clustering to be examined for different plant features.

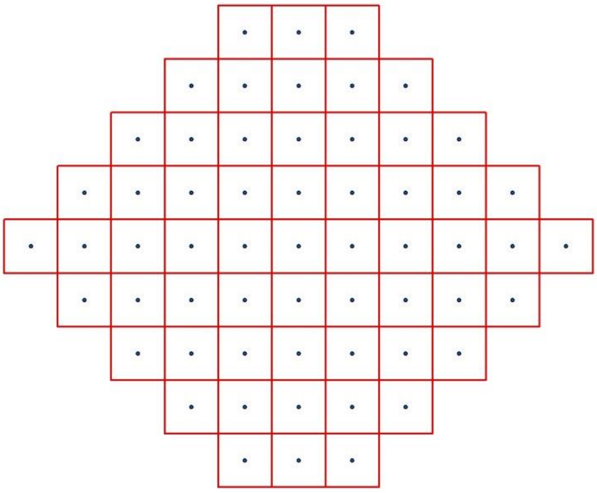


Fig. 4.3. 'Circular' grid of 59 cells surrounding planting spaces used to record habitat features.

4.3.7 Microhabitat variables

Twelve microhabitat variables were extracted from the microhabitat survey data in seven categories (Table 4.1). These variables were chosen to test a-priori hypotheses on the drivers behind species richness and features that have been identified previously in other studies. For each hypothesis, the variables selected were also those that reduced the correlation with other variables as much as possible ($r_{Pearson} < 0.6$; although see the one exception below), in order to minimise the effects of multi-collinearity.

Table 4.1. Descriptions and ranges for the microhabitat variables

Category	*Variable*	*Description*	*Range*
Vertical structure	UnderOverH	Evenness of overstorey and understorey plants, reflected by the Shannon-Weiner diversity index (H)	0 - 0.7
Horizontal structure	ANN Plants	Average nearest neighbour index for plants, representing level of clustering of plants	0.8 - 1.6
	ANN Understorey	Average nearest neighbour index for understorey, representing level of clustering of understorey	0.4 - 1.9
Specific features	Fallen Dead	Fallen dead trees per hectare, reflecting large fallen timber (logs)	0 - 24
	Dead Trees	Dead standing trees per hectare	0 - 52
Ground	GroundH	Diversity of ground substrates as represented by the Shannon-Weiner diversity index (H)	0.6 - 1.8
Floristics	OverstoreyH	Diversity of overstorey plants, as represented by the Shannon-Weiner diversity index (H)	0 - 2.5
	UnderstoreyH	Diversity of understorey plants, as represented by the Shannon-Weiner diversity index (H)	0 - 1.6
	Local Plants	Plants of species found in the Mount Lofty Ranges per hectare	0 -143
Plant density	Plant Density	Density per hectare of all trees and large shrubs > 1 m tall (excluding recruits)	88 - 293
Site scale	DistWoodRem	Distance to patches of woodland remnant > 1 ha in metres	26 - 2087
	DistGrazedRV	Distance to patches of grazed revegetation outside site in metres	37 - 1083

To represent vertical structure, plants were grouped into understorey – those species that provided low dense foliage (i.e. shrubs, and *Callitris* and *Pinus* spp.), and overstorey – tree species that lacked low dense foliage (i.e. most eucalypts). Initially, both these variables were to be included separately, and together with an interaction term, but were highly correlated ($r_{Pearson}$ = -0.6). Instead, the evenness of understorey and overstorey (UnderOverH) was calculated using the Shannon-Weiner diversity index (H), to test the hypothesis that a mix of trees and shrubs facilitated greater species richness.

Variation in horizontal structure was incorporated to reflect the theory that every bird species requires a certain vertical foliage profile, and that an area of habitat containing a range of patches with different foliage profiles will support a greater diversity of bird species (*sensu* MacArthur *et al.* 1962). For woodland systems, it was assumed that generally bird species could be grouped into those that require understorey (denser foliage profiles; e.g. Fairy-wrens, Babblers), and those that require more open areas without understorey (more open foliage profiles; e.g. robins, flycatchers, woodswallows), and therefore an area that provides patches of both would support more bird species. Horizontal structure was calculated using the Average Nearest Neighbour (ANN) index in ArcGIS - a measure of the clustering of features based on the ratio between the observed mean distance between each plant and their nearest neighbour, and the expected mean distance given a random pattern (ESRI 2008). Two variables were selected, the first – ANN Understorey, was a measure of the degree of clustering of understorey, and was designed to represent the presence of both patches of understorey and more open areas below or around overstorey. The second variable - ANN Plants, was a measure of the degree of clustering of all plants, and was chosen to determine if the presence of open spaces or clearings (the most open foliage profiles, devoid of both understorey and overstorey) were also important in contributing to bird diversity (i.e. the more clustered the plants the more open space).

Two specific habitat features, Fallen Dead and Dead Trees, were selected due to their potential influence on multiple species. Fallen Dead corresponded to the number of fallen dead trees, representing large fallen timber, a feature shown to be important for many woodland bird species (Mac Nally & Horrocks 2007). Similarly, standing Dead Trees have been considered as a key predictor in similar studies (e.g. Loyn *et al.* 2007, Mac Nally *et al.* 2010), and have been used heavily by several species in this system (e.g. Varied Sittella, Tree Martin and three species of Woodswallows; unpublished data).

Ground layers were represented in a single diversity index (using the Shannon-Weiner method), designed to test the hypothesis that a variety of ground foraging birds would be enhanced by increased diversity of ground substrates. The dominant layers such as litter, bare ground, and small shrubs, were not included individually as they were all highly correlated themselves or with other non-ground variables (Appendix 11).

Floristic diversity was represented through two diversity indices: the diversity of overstorey species (OverstoreyH), and the diversity of understorey species (UnderstoreyH). These two indices were used instead of an overall diversity index for all plants, because of concerns that plant diversity would also represent structural diversity (i.e. adding species of understorey to an area of overstorey or vice versa would also represent an increase in the vertical structural diversity). Another floristic variable, the number of local plants, was selected to test the theory raised by other authors that local plant species provide better habitat than non-local species (Barrett & Davidson 1999, Bennett *et al.* 2000).

Plant density was included because it has been shown to have an influence on several tree-borne resources used by woodland birds (Vesk *et al.* 2008), and because it is an essential component required by revegetation practitioners. The current density of plants was used rather than the original planting density, as many plants died in the first few years after establishment (SA Woods & Forests unpublished data), and therefore would not have had any competitive influence on the surrounding plants. For the same reason, plants less than 1 m tall that had been stunted or failed to grow well, were not counted, in addition to recruits (regenerating plants), most of which were less than 2 m tall and had only established recently. Plant density was highly correlated with GroundH ($r_{Pearson}$ = -0.63 and $r_{Pearson}$ = -0.73 for the All Species and Declining Species datasets respectively), but both variables were retained as it was thought that each would explain the presence of certain resources better than the other (e.g. plant density for canopy level resources, and ground diversity for ground layers potentially unrelated to plant density such as small shrubs).

Finally, two site scale variables (DistWoodRem and DistGrazedRV) were also included to account for any variation in species richness caused by relevant broader scale factors. DistWoodRem represented the distance to patches of woodland remnant > 1 ha and was designed to account for species that have been shown to use the revegetation more often in conjunction with woodland remnant (e.g. Brown Treecreeper, Southern Whiteface; Chapter 2, Barker 2007, Hoffmann 2011). Similarly, DistGrazedRV reflected the distance to patches of grazed revegetation, and was included to factor in the positive effect this feature has on the relative abundance of several declining species (Chapter 2). These variables were only designed to account for variation in species richness between plots within individual sites. Variation in relation to broader differences in the landscape context of sites was taken into account during the analysis (see 4.3.9 below).

4.3.8 Response variables

The response variables were the species richness values for all woodland bird species (herein All Species) and declining woodland bird species (herein Declining Species) at each of their sample locations, which were derived from the respective distribution surfaces. The raw number of species recorded in the distribution surfaces was used rather than the five classes used to stratify the selection of sample locations, in order to maintain any informative variation within classes.

4.3.9 Analyses

Microhabitat modelling

To determine the influence of the microhabitat variables on species richness, generalised linear mixed models (GLMMs) were constructed for each response variable. GLMMs combine the ability of generalised linear models (GLMs) to handle non-normal data (e.g. count, binary), and the ability of mixed models allow for population level inference in datasets based on grouped sampling designs through the use of random effects (Bolker *et al.* 2009). Here, site was used as a random effect to cater for the grouped design of samples within survey sites, and account for any differences in landscape context in the plots between sites. As richness values constitute count data, GLMMs with Poisson errors were used, and these were fitted using the 'glmer' function in the 'lme4' package (Bates *et al.* 2011) in R (R Core Team 2011). In addition, GLMMs for both responses were fitted with individual level random effects to account for overdispersion using Plot ID as the random effect (Bates *et al.* 2011).

As there were potential multidimensional effects of the microhabitat variables on species richness but no clear combinations of variables, all possible models were evaluated using the 'dredge' function in the 'MuMIn' package for R (Barton 2011). Models were compared using the small sample correction of Akaike's Information Criterion (AIC_c), and the derivatives: delta AIC_c (Δ_i) – the relative difference between models and the model with the lowest AIC_c, and Akaike weights (w_i) – the relative likelihood of the model in the candidate set of models (Burnham & Anderson 2002).

To account for model selection uncertainty and assess the importance of each individual variable, model averaging was conducted on models with reasonable support from the data ($\Delta_i < 7$; Burnham & Anderson 2002) using the 'model.avg' function from the 'MuMIn' package. Model averaging provided averaged regression coefficients (ARC; the average effect of each parameter across all models), and the sum of the Akaike weights (Σw_i), indicating the relative importance of each variable (RVI) in a standard index between 0 and 1 (Burnham & Anderson 2002). To avoid biasing coefficients away from zero, full model averaging was used to calculate the ARC and corresponding standard error, with each variable included in every model and coefficients set to zero in models where a parameter did not appear (Barton 2011). Key predictors were regarded as those that had high RVI and standard errors of full model-averaged regression coefficients that did not include zero (*sensu* Haslem & Bennett 2008).

Model adequacy

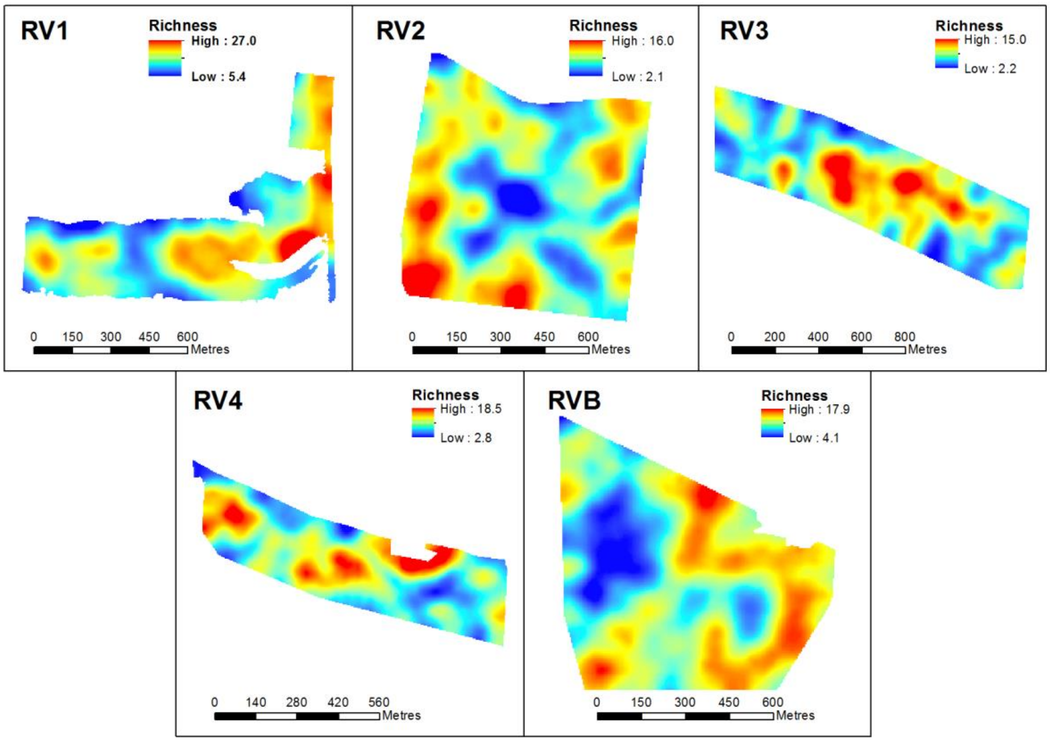
To evaluate the effectiveness of the microhabitat variables selected, the goodness-of-fit was calculated for each response variable. As standard goodness-of-fit coefficients (R^2) cannot be generated for mixed models, a goodness-of-fit measure specifically adapted for generalised mixed models (R^2_{GLMM}) was used (Nakagawa & Schielzeth 2013). This involves comparing the full model (model containing all fixed and random effects), and the null model (model containing only the intercept and random effect). Two statistics are calculated: the marginal regression coefficient ($R^2_{GLMM\ (m)}$), representing the variation explained by the fixed effects; and the conditional regression coefficient ($R^2_{GLMM\ (c)}$), representing the variation explained by fixed and random effects. Here, the marginal regression coefficient ($R^2_{GLMM\ (m)}$) was used to estimate the variation explained by the microhabitat variables.

4.4 Results

4.4.1 Distribution surfaces

The richness of both All Species and Declining Species varied greatly within all five survey sites (Fig. 4.4). All sites contained very apparent ‘hot’ areas that were used by up to 27 species and conversely ‘cold’ areas that were used by less than six species. The differences between these areas were large with 4-8 fold differences in species richness for All Species and 7-11 fold differences for Declining Species.

a)



b)

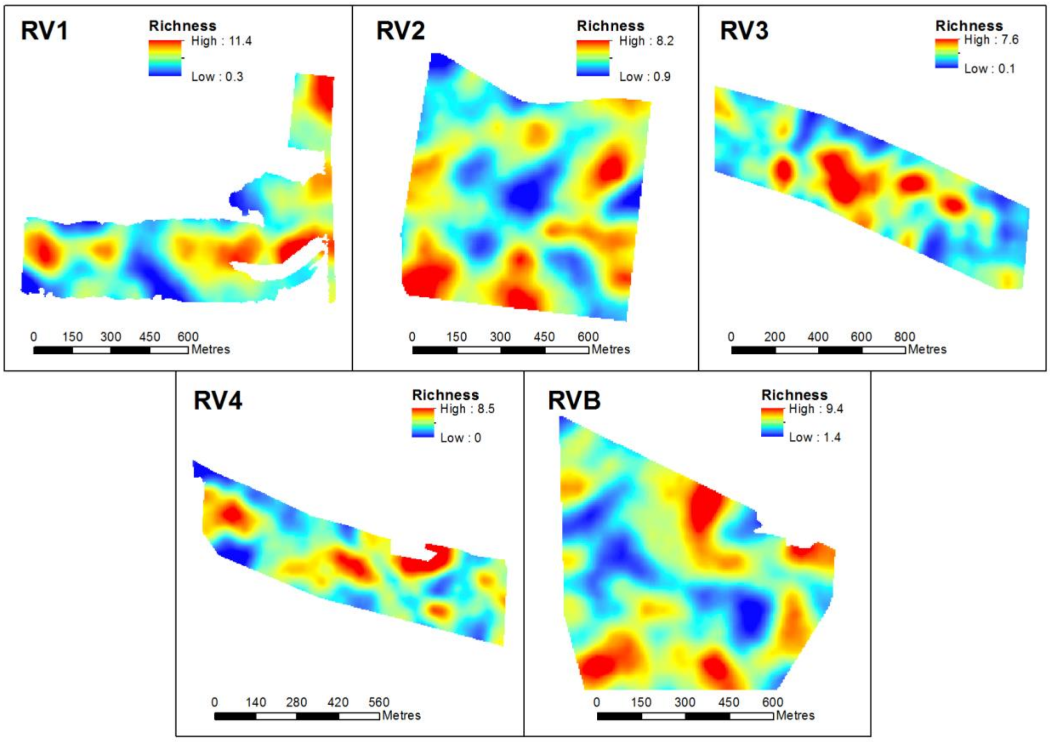


Fig. 4.4. Variation in species richness in the five survey sites for: a) All Species, and b) Declining Species. Images are relative to the species present in each site, and therefore are visualised on separate richness scales. Site names are indicated in the top left hand corner of each map.

4.4.2 *Microhabitat modelling*

For each response variable, a large number of models with reasonable support from the data were selected (All Species = 589, Declining Species = 255). None of the models however were clearly supported above the others (Akaike weights ± 0.01), and all of the models selected had weights < 0.1, well below the threshold of 0.9 required to accept a single best model (Burnham & Anderson 2002). Furthermore, there was no consistency in the top models selected between the response variables.

Model averaging on the other hand, revealed clear differences between the microhabitat variables (Table 4.2; Fig. 4.5). Across both responses four variables had ARC standard errors that did not include zero and were considered key predictors. OverstoreyH was the most important being the top ranked variable for both responses according to the RVI. For All Species, this was followed by GroundH and UnderOverH respectively (Fig. 4.5a). In contrast, for Declining Species these two variables were relatively unimportant and instead Plant Density was the second most important variable (Fig. 4.5b). The ARC for this variable was negative indicating there were higher numbers of declining species at lower densities of plants.

Of the remaining variables, both clustering indices (ANN Plants and ANN Understorey) were relatively unimportant. ANN Understorey did have a moderately high RVI of 0.53 in the All Species analysis but had a highly variable ARC (0.1995 ± 0.2403), suggesting it was not consistently important. In terms of the floristic variables, unlike OverstoreyH, UnderstoreyH showed comparatively little influence across both responses. Like ANN Understorey though, it had a moderately large RVI of 0.48 for All Species although this was again highly variable (0.1007 ± 0.1377). The other floristic variable, Local Plants was also unimportant being ranked amongst the lowest variables for both responses. Similarly, the specific features of Fallen Dead and Dead Trees were also amongst the lowest ranked, suggesting they had little influence on either response. Finally, both the site scale variables, had very low ARC (< 0.0001) indicating their effects on the responses were small, although DistGrazedRV and DistWoodRem had moderate RVI values for All Species and Declining Species respectively, suggesting they did play a role in the associated analyses.

Table 4.2. Averaged regression coefficients (ARC) ± Standard Errors (SE), and Relative Variable Importance (RVI; Σw_i) values for each microhabitat variable in the two response variables. ARC and SE were calculated from a full model-average. Values in bold indicate variables with RVI values > 0.7 and ARC > SE.

	All Species		*Declining Species*	
Microhabitat variables	ARC	RVI	ARC	RVI
UnderOverH	**0.6590 ± 0.5548**	**0.72**	-0.1005 ± 0.3208	0.21
ANN Plants	-0.1197 ± 0.2588	0.31	-0.0406 ± 0.2233	0.17
ANN Understorey	0.1995 ± 0.2403	0.53	0.0912 ± 0.2090	0.28
FallenDead	0.0028 ± 0.0069	0.25	-0.0002 ± 0.0041	0.14
DeadTrees	0.0005 ± 0.0023	0.16	-0.0001 ± 0.0027	0.14
GroundH	**0.6681 ± 0.3746**	**0.88**	0.0538 ± 0.2311	0.21
OverstoreyH	**0.2780 ± 0.1550**	**0.89**	**0.3259 ± 0.1601**	**0.95**
UnderstoreyH	0.1007 ± 0.1377	0.48	0.0113 ± 0.0672	0.16
LocalPlants	0.0007 ± 0.0013	0.36	-0.0005 ± 0.0015	0.24
Plant density	0.0003 ± 0.0010	0.22	**-0.0042 ± 0.0025**	**0.88**
DistWoodRem	-0.0002 ± 0.0002	0.25	0.0000 ± 0.0001	0.39
DistGrazedRV	0.0000 ± 0.0001	0.54	0.0001 ± 0.0001	0.14

4.4.3 Model adequacy

For All Species, the marginal regression coefficient indicated the microhabitat variables selected explained just over half of the variation in the data ($R^2_{GLMM\ (m)} = 0.52$). On the other hand, the marginal regression coefficient for the Declining Species analysis suggested the microhabitat variables explained just under a quarter of the variation ($R^2_{GLMM\ (m)} = 0.24$).

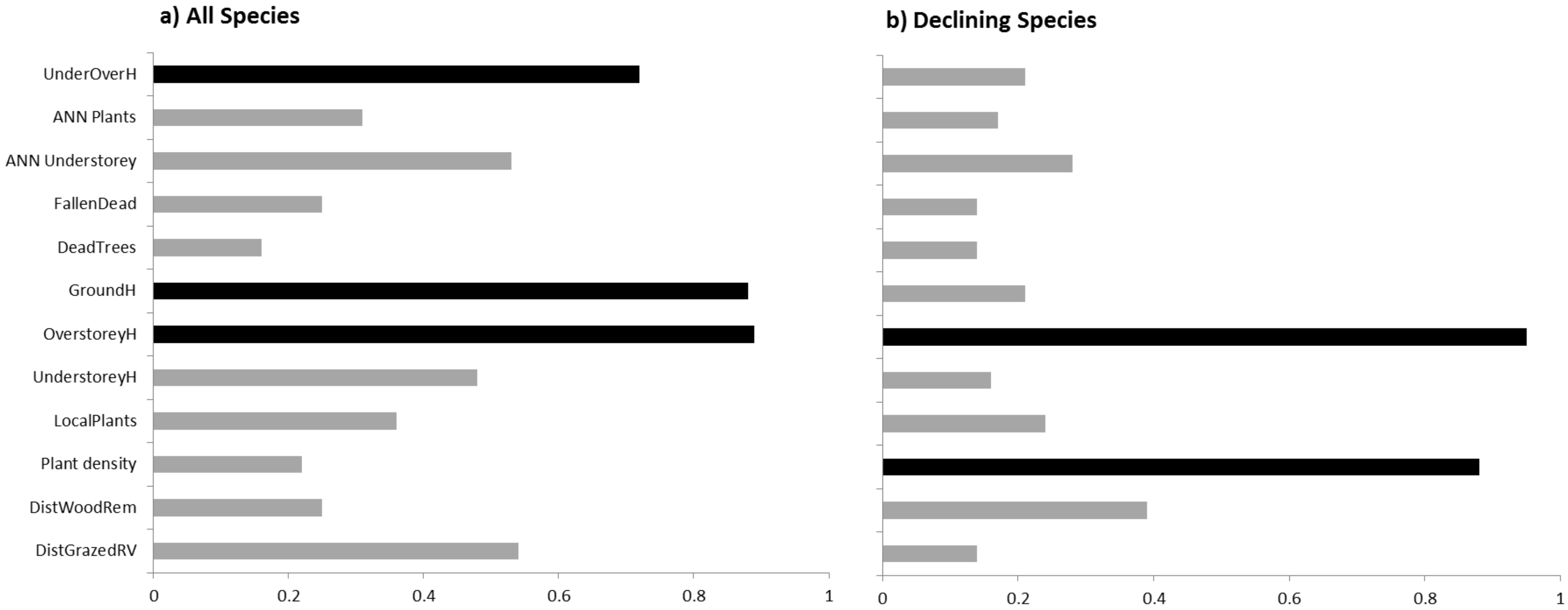


Fig. 4.5. Relative Variable Importance (Σw_i) of the 12 microhabitat variables for each response variable. Black bars indicate variables where the standard error of the full model-averaged regression coefficients did not overlap zero.

4.5 Discussion

4.5.1 Key microhabitat features

The diversity of overstorey plant species was the most important microhabitat feature in explaining the patterns of richness for both All Species and Declining Species. This result is not surprising, because numerous bird species have been shown to exhibit strong preferences for particular tree species which are often distinct from similar species occupying the same area (e.g. Holmes & Robinson 1981, Recher & Majer 1994, Gabbe *et al.* 2002). Therefore, logically, as has been shown, an increase in the number of tree species can lead to higher numbers of bird species (e.g. Peck 1989, Matlock & Edwards 2006, Gil-Tena *et al.* 2007). Interestingly though, despite this well-established relationship, floristic diversity and particularly overstorey diversity, has not often been tested in studies of birds in revegetation (e.g. Selwood *et al.* 2009, Lindenmayer *et al.* 2010, Mac Nally *et al.* 2010). Moreover, of those studies that have included this variable, all but one (Gardali & Holmes 2011) found structural effects of the vegetation rather than floristic diversity were important (e.g. Twedt *et al.* 2002, Barrett *et al.* 2008, Munro *et al.* 2011). The influence of overstorey diversity here reinforces the importance of floristic diversity, particularly tree species diversity, as a key microhabitat feature and a useful mechanism to increase the number of bird species in reconstructed habitat.

For All Species, the second most important feature was ground diversity, which is also logical as around a third of all the woodland birds recorded are ground foragers, and previous research has suggested different ground layers are required to support different ground foraging species (Antos & Bennett 2006, Antos *et al.* 2008). The result here supports these findings and highlights the need to include the full range of ground layers in reconstructed habitat in order to provide for a variety of ground foraging species. Unlike overstorey diversity though, ground diversity does not directly translate to management and an understanding of the underlying mechanisms will be required in order to recreate it. Grazing at low to moderate levels may be a partial solution, as it has been suggested that grasses are kept from dominating other ground layers without being wiped out, and that this may explain the increase in use of ground foraging birds at these levels (Maron & Lill 2005, Martin & Possingham 2005). However, while important, grazing will not enhance the development of features primarily borne by overstorey plants like fallen timber and litter. Hence, the exact drivers behind the most diverse areas containing these features as well may need more research. In the meantime though, general guidelines can be postulated. For example, logically areas around plants will contain more litter, and those further away from plants will likely contain more bare ground and grass, and therefore providing areas with and without plants may be a way to achieve greater ground diversity. Also, fallen timber has been shown to be more prevalent in lower density plantings (Vesk *et al.* 2008), and this may be another way to contribute to higher ground diversity (see below). The relative proportions of each substrate and the optimal size of the patches required though is unknown, and further research on ground foraging birds will be needed to answer these questions. Until then, incorporating ground layer diversity through even a minor degree of spatial variation in planting, and possibly complementing this through grazing at low levels, should at least increase the chance of more ground foraging species using reconstructed habitat.

The third important feature for All Species was another diversity index indicating the evenness of overstorey and understorey plants, although unlike the previous key features this demonstrates the need for structural diversity rather than diversity of resources. As mentioned earlier, a number of studies have shown vegetation structure is an influential factor on birds using revegetation, with cover in several vertical layers demonstrated to increase the richness and abundance of bird species (Barrett *et al.* 2008, Lindenmayer *et al.* 2010, Munro *et al.* 2011). Although the variable here is a much simpler version (presence of two broadly defined layers – trees and shrubs), it is consistent with these findings, and highlights the need to include a mix of overstorey and understorey plants to cater for a variety of woodland birds in reconstructed habitat. However, while this suggests a relatively even mix of trees and shrubs, it does not necessarily suggest the need for these to be spatially even. This is important, because some species of ground foraging woodland birds require areas without shrubs in order to access the ground (Antos *et al.* 2008), and even though the clustering variables were not important here, this suggests there should be spatial variation in the establishment of trees and shrubs. In this way, both species that require areas with and without shrubs can be provided for. Further research though will be required to determine how large such areas need to be.

Lower plant density was the final key microhabitat feature identified, although this was only important for Declining Species. Such a finding is not unexpected, as a number of declining species have been shown to exhibit a negative response to the density of trees and shrubs (Antos *et al.* 2008). In addition, as mentioned previously, lower densities of plants have been linked to increased levels of fallen timber (Vesk *et al.* 2008), which is a substrate that many declining species use (Antos & Bennett 2006, Antos *et al.* 2008). Moreover, lower densities are also thought to enhance the development of low branches (Paton *et al.* 2004), which are used as foraging platforms by several species of declining ground pouncing woodland birds (Recher *et al.* 2002, Gillespie 2005, Antos & Bennett 2006). This result supports these findings, and shows that the benefits of lower densities to individual species, does in fact translate to multiple species using the same area and therefore is an important mechanism for providing suitable habitat for a variety of declining woodland birds.

Furthermore, lower plant densities may also contribute to the development of higher ground diversity, and this may explain why ground diversity was not important for declining species even though over half of these species are ground foragers. As discussed earlier, open spaces may increase ground diversity and naturally open spaces will be more common at lower densities of plants. Hence, lower plant densities may have explained some of the variation in ground diversity for declining species (indeed, these variables were negatively correlated, particularly in the Declining Species dataset: $r_{Pearson} = -0.73$; Appendix 11). This relationship will need to be tested explicitly, but if significant, means that lower plant densities can be used to increase the number of declining species, not only through the provision of tree-borne resources such as low branches, but also through a mix of ground layers. An obvious question that arises from this result though, and one that has not yet been addressed by other authors, is exactly how low does the density of plants (i.e. trees and large shrubs) need to be in order to achieve these effects? Exploratory analyses of the fitted values indicated there was no threshold and the highest predicted richness was at the lowest plant density recorded of 88 plants ha^{-1}. Therefore, further research incorporating lower densities will be needed to determine how much lower the density should be. Until then, it can be assumed based on the data here, that a density of < 100 ha^{-1} will be needed, at least in woodland systems similar to Monarto.

4.5.2 Management implications

Together, these results indicate that to provide suitable habitat for a range of typical and declining woodland birds reconstructed habitat needs to include a mix of overstorey and understorey plants, comprised of a range of overstorey species, planted at low densities and incorporating a variety of ground substrates. Current plantings however, often demonstrate the opposite. For example, Harris (1999) examined the characteristics of plantings around the Tungkillo region in the eastern Mount Lofty Ranges and found that most were comprised of only a few tree species, on average had double as many trees than shrubs, and were planted at high densities only a couple of metres apart. Obviously, such methods will need to change considerably if suitable habitat for woodland birds is to be created. One component that may be difficult to change though, is the inclusion of more overstorey plant species, as a goal of many revegetation programs (particularly those aimed at re-establishing habitat) is to recreate the plant communities that are believed to have formerly occupied the area (Miller & Hobbs 2007). This may be a problem, as most pre-European vegetation communities are only classified as having 1-2 species of overstorey plants (e.g. *Eucalyptus porosa* woodland, or *Eucalyptus leucoxylon* and *Eucalyptus camadulensis* open forest; Kraehenbuehl 1996). The results presented here though, strongly suggest that if suitable habitat for a range of woodland birds is to be provided in the limited opportunities available, then practitioners need to move beyond this view of habitat and incorporate other suitable overstorey plants.

4.5.3 Improvements

While these features were clearly identified as important, there was still a large amount of unexplained variation in the analyses for both responses, and therefore some improvements may be needed to ensure none of the other variables were also important. First, the scale at which the microhabitat features were sampled (ca. 25 m radius) may have been insufficient to capture all the relevant variation in some of the features. For instance, during sampling it was noticed that due to the high variability in the mix of plant species established throughout the plantings (e.g. Fig. 4.1), even within an area of consistent richness there was considerable patchiness in different features and the sampling plots did not always capture this variation (J. Allan personal observation). In particular, plots were often positioned in an area of either high numbers of overstorey or understorey plants, or an area of generally high or low plant density, immediately adjacent to areas of the opposite structure, and this may have reduced the effect of the clustering variables. Sampling at the scale used to calculate the richness (50 m radius) is the logical solution, and initially this was the plan, but as mentioned was infeasible due to the detailed vegetation measurements collected. Hence, reducing the detail of the sampling would be necessary to sample at this scale, and one possible way to achieve this would be eliminating the ground cover variables which were the most time consuming aspect of the data collection. As mentioned, whilst very important, the ground layers are primarily a function of plant characteristics, and therefore could be approximated through plant measurements without being recorded directly. This would enable more rapid assessment of microhabitat features and the collection of data across larger areas more feasible.

Second, while the bird data here come from a large number of surveys spread across multiple seasons, due to time constraints they were confined to only one year, and the richness values therefore may not be completely reflective of the use of species over longer timeframes. For example, it is possible that annual fluctuations in resources or species presence may have meant that some areas could have had more species using them than recorded here, and accounting for this may clarify some of the variability surrounding some of the microhabitat features. Flowering resources in particular were highly variable between the survey period and preceding years, with several species of eucalypts flowering extensively prior to surveying but not during the survey period and vice versa (unpublished data). Spatial variation in the planting of different eucalypt species may therefore have resulted in an inaccurate reflection of the richness of nectarivorous species in some areas. Data on 100s of flowering plants were collected during each survey season in order to account for such variation but this was not adequate to capture the variability in flowering shown. Therefore, ultimately in order to capture these and other annual variations, surveys will need to be extended to incorporate multiple years.

Finally, it is possible that the bird distributions displayed here, are not only a function of the characteristics of the vegetation, but also influenced by potential differences in underlying productivity within the sites. For instance, revegetation associated with more productive areas along drainage lines influenced bird distribution at the landscape scale (Chapter 2), and similar effects could be occurring at finer scales within sites, not just to drainage lines but other fine scale differences in soil and topography. If fine-scale maps of soil and topographic characteristics were able to be sourced, then as with the site scale variables, these could be added to the analysis to help improve the variation explained and further clarify the effects of the microhabitat variables.

4.5.4 The next steps

Once these improvements are made, the approach may also need to be expanded to increase its practical applicability and representativeness. For instance, while the identification of key microhabitat features is an important step, ultimately predictive models – models that discern the specific amounts of features required will be needed to fully inform habitat reconstruction. The approach used here is particularly suited to developing accurate predictive models, because it eliminates variability added through sampling features that correspond to another level of habitat usage (i.e. a randomly placed survey plot overlapping both a high and low use area), and should therefore enhance the ability to identify the optimal amounts of features needed. The reason predictive models were not developed here, was firstly because, as mentioned above, there was still a large amount of variation unexplained by the analysis, which raised concerns over the accuracy of any subsequent predictions. Also, many of the variables, especially the top variables, cannot be directly translated to management (e.g. the diversity and clustering indices). These variables would need to be converted to more practically meaningful measures (e.g. richness instead of diversity and area of clusters or spaces instead of index values) in order to make useful predictive models. Once these improvements are made however, this process can be used to create predictive models that have increased confidence in the accuracy of their predictions.

Another step that may need to be taken is the development of models for individual species, as although assessing microhabitat features using species richness gives an idea of the requirements of multiple species, it does not necessarily represent the requirements for every individual species. This is possible, because areas of high richness may not directly coincide with high use areas for individual species, and this could lead to important effects for certain species not being detected. Furthermore, even if these areas coincided, factors unique to an individual species could be missed simply because they are not used by other species. Therefore, ultimately individual species models will be needed to give the best possible chance of any given species occupying reconstructed habitat. The difficulty in developing individual species models with this approach though, is that in order to ensure sample plots correspond to a consistent level of use, microhabitat sampling regimes for every individual species need to be implemented, as most of their distributions will inevitably be at least slightly different. Originally, the idea was to completely sample all of the microhabitat features within each survey site, and then sample for each individual species from that dataset - as has been done when this method has been used previously in other parts of the Mount Lofty Ranges (e.g. Mt Bold, Para Woodland; Paton unpublished). However, the large size of the survey sites combined with the detailed vegetation measurements here forced features to be subsampled. Logistically, microhabitat sampling regimes for only two response variables could be implemented, and rather than sample microhabitat features for only two individual species and ignore the rest, these were chosen to be measures of species richness. If significant reductions in the sampling logistics such as those mentioned previously were made however, then data on microhabitat features could foreseeably be obtained over the whole of each site, and individual species models could then be developed using the same process implemented here. These models could then be used in conjunction with models developed for multiple species to give the best chance of catering for individual species, while at the same time increasing the potential for overall diversity of woodland birds.

Even with these changes though, this method may not be able to adequately represent the use of some less common individual species. This may be the case, because each area search is only designed to collect one point representing the location of each individual bird or group of birds, and for those species whose abundance is not sufficient to be encountered at least once in the areas they frequently use during the survey periods, then this may not provide an adequate representation of their microhabitat requirements. For instance, the distribution of Red-capped Robins at one of the survey sites obtained by this method missed large areas that were frequently used by this species when compared to the distribution obtained in a project targeting this species during the same period (Appendix 12). Theoretically, a more adequate distribution could be obtained by simply increasing the number of surveys, and eventually detecting a bird in most of the areas they use frequently. However, to obtain such distributions many individual species studies collect at least 50 points per bird (e.g. Barg *et al.* 2006, Anich *et al.* 2012), which would mean four times the survey effort here and more if the bird was not detected in every survey. Obviously, such effort would be infeasible and hence this method may also need to move to individually targeting the less common species in home range style studies to ensure their microhabitat requirements are adequately represented. This will be important as many of the less common species are not surprisingly also those that are declining, and making this step will help ensure reconstructed habitat is able to cater for these as well as more common woodland birds.

4.5.5 Conclusion

This study has for the first time, identified key microhabitat features for reconstructed habitat based on the fine scale distribution of habitat use. As this method enabled the identification of features that characterise the exact locations used, there can be increased confidence that some of the finer variation in habitat requirements has been captured. In turn, when combined with existing results incorporating variation at broader scales, this can give much greater confidence that any resulting reconstructed habitat will support the species concerned, which is crucial when there are limited opportunities for habitat reconstruction.

Chapter 5
General Discussion

5.1. Overall implications

Overall the results presented in this thesis suggest that in order to provide for a range of typical and declining woodland birds reconstructed habitat should be established along drainage lines in large blocks, 100s of hectares in size, and incorporate a diversity of overstorey plant species intermixed with understorey plant species, planted at low densities and including a variety of ground layers. Together these results point to the need to be highly strategic when planning reconstructed habitat: concentrating efforts in large plantings in specific areas, and carefully considering the mix and density of plants. However, as noted by other authors, most current revegetation programs illustrate the opposite, being conducted in a piecemeal manner from the ‘bottom up’ with little coordination (Bennett & Mac Nally 2004, Mac Nally *et al.* 2010, Paton *et al.* 2010b). For example, most plantings are small, isolated and situated on whatever land property owners are willing to give up (Bennett & Mac Nally 2004), and at the plant level while general planting guidelines are provided (e.g. Corr 2003), decisions are ultimately left to the discretion of the practitioner. The results presented here suggest this practice needs to change if biodiversity is to be conserved with the limited opportunities and resources available, and these results can help form the framework required for such a change.

5.2. Future improvements

While the approach used here has successfully provided a range of guidelines for habitat reconstruction, there are a number of limitations that could be improved in future. Some of these such as improving temporal replication and developing predictive models have already been discussed in the preceding chapters, but there are two other potential limitations that have not yet been discussed in detail.

First, although the general Monarto region is similar to other woodland areas, certain features of the revegetation are unique and subsequently the broader applicability of some results may need to be investigated. For instance, the effects of an extremely high level of floristic diversity that is unnatural and unlikely to be repeated, is unknown, but could for example lead to greater resources and smaller area requirements compared to other reconstructed areas. Also, other characteristics such as the low planting density and great size, shape and aggregation of the plantings may have been responsible for its success, but the overall bias towards these extremes could have meant these factors were not identified as important when in reality they are. Hence, this approach should be expanded to other areas to test exactly how general the findings are.

Second, all of the guidelines were based only on measures of occupancy (presence or abundance). As noted by other authors, this could be an issue because occupancy does not necessarily equate to persistence, and ultimately information on breeding requirements are needed to ensure species are able to survive in the long term (Loyn *et al.* 2007, Selwood *et al.* 2009, Mac Nally *et al.* 2010). This does not devalue the results presented here though, as they provide the baseline required for species to be able to use an area, and breeding requirements will simply add another layer of understanding on top of this. Also, there is evidence that many woodland bird species, including most of the declining species recorded, are breeding in the revegetation at Monarto (Paton *et al.* 2004; unpublished data). Therefore, the results obtained here are likely to be related to breeding requirements. Nonetheless, these will not account for any differences in breeding in regard to different habitat features throughout the revegetation and in future this approach should be expanded to incorporate breeding requirements. For example, using breeding evidence (*sensu* Selwood *et al.* 2009, Mac Nally *et al.* 2010) in birds recorded in landscape surveys to assess landscape features, using the fine-scale distribution of breeding birds to determine key microhabitat features, and using home ranges with successful breeding to ascertain the minimum areas required. In doing so, this approach can then be used to capture some of the finer variation in breeding not just habitat use, and produce robust guidelines that increase the chance of both species occupancy and persistence.

5.3. Wider application of approach

Given guidelines were able to be produced using methods that closely reflect habitat use, this approach could be used for similar work in future. However, as discussed in the preceding chapters, this approach is much more intensive than traditional plot and patch based methods, and this may limit its use, particularly in the context of increasing spatial and temporal replication as mentioned in the previous section. It should be noted though, that some methodological aspects unrelated to the approach may have exaggerated its intensiveness here. For example, as mentioned previously, in hindsight some microhabitat features (e.g. ground layers) did not need sampling in as much detail as they were, and the frequency of fixes used to generate home ranges and determine area requirements could also be greatly reduced. Similarly, the landscape sampling incorporated remnant vegetation which while important for assessing the success of the plantings, more than doubled the sampling effort and yet was unnecessary for the purposes of this thesis. Eliminating these aspects would greatly reduce the intensiveness and likely have allowed more robust levels of spatial and temporal replication during this study, i.e. greater sampling coverage of microhabitat, more home ranges for estimating area requirements and multiple years of sampling for assessing landscape features. Therefore, any future proponents of this approach should take this into account when assessing its feasibility.

Even with these changes though, the approach used here will still be more intensive than its traditional plot and patch based counterparts, and therefore this may still limit its applicability. A possible solution however, could be to use it as a complementary approach. For instance, the traditional approach could be used as the main sampling technique for most work across a given region, and the approach used here could be used to target those species unable to be adequately represented by the main technique (e.g. uncommon species less likely to be detected, or species with subtle requirements), or as a check of the main results in a small part of the region. In this way, the best of both approaches would be gained: more feasible sampling over broader spatial and temporal scales from the traditional approach and a comprehensive assessment of habitat requirements provided by the approach used here.

Ultimately though, exactly how feasible the approach used here is and how it should be used, will depend on the magnitude of the benefit it provides compared to traditional approaches, in terms of understanding habitat requirements and the ability to recreate suitable habitat. Discerning this information will necessarily require a direct comparison of both approaches. For example, as mentioned previously, in their work on the habitat use of warblers, Barg *et al.* (2006) and Anich *et al.* (2012) simultaneously compared results gained from sampling that accounted for spatial variation in habitat use within territories, to those from sampling at the territory level that did not take this variation in to account. Their work showed that accounting for this variation provided a more detailed understanding of habitat requirements, highlighting habitat features critical to the species life history. In hindsight, such a comparison could have also been implemented in this thesis or even established beforehand, however the focus at the time was on answering the practical questions and at least for the landscape and microhabitat levels would have also required more sampling (i.e. implementing associated plot or patch based sampling regimes) which would have been infeasible in the time available. Future research though, should consider undertaking this comparison. Such a comparison would also allow value of information analysis to be performed (e.g. Runge *et al.* 2011, Maxwell *et al.* 2015), which evaluates the benefit of gaining more information relative to undertaking other approaches or management actions, in terms of the improvement in management performance it provides (i.e. increased chance of species occupancy), and can therefore be used to determine exactly how much improvement in recreating suitable habitat would likely be gained for the increased effort. Until then, researchers should make their own decision whether to use this approach based on their knowledge of the spatial variation of the study system and the species in question, and the ability of traditional approaches to capture all the relevant variation.

5.4. Conclusion

Declines in biodiversity - particularly woodland birds, are ongoing, suggesting that species will disappear if suitable habitat is not reconstructed over the next few decades. The results presented in this thesis represent a range of important habitat features for woodland birds that can be used to enhance the effectiveness of any future reconstructed habitat from the landscape down to the plant level. As these features were determined using a detailed, focussed, organism-orientated approach, there can be increased confidence that some of the finer details of habitat requirements have been captured. These results can therefore be used to complement existing results incorporating broader variation, and reinforce those where they overlap. Importantly, this also provides increased confidence that in the face of extinction and in the limited opportunities available, reconstructed habitat will indeed be successful in supporting the species concerned, and ultimately at ensuring their persistence for future generations.

Appendices

Appendix 1. List of 81 bird species recorded in the revegetation for Chapter 2, with their broad habitat categorisation, and declining status in the Mount Lofty Ranges according to Paton *et al.* (2004). Species are listed in taxonomic order.

Common name	Species name	Broad habitat	Declining?
Emu	*Dromaius novaehollandiae*	Open country	
Stubble Quail	*Coturnix pectoralis*	Open country	
Painted Button-quail	*Turnix varia*	Woodland	Yes
Straw-necked Ibis	*Threskiornis spinicollis*	Open country	
Black-shouldered Kite	*Elanus axillaris*	Open country	
Little Eagle	*Hieraaetus morphnoides*	Open country	
Brown Goshawk	*Accipiter fasciatus*	Woodland	
Collared Sparrowhawk	*Accipiter cirrhocephalus*	Woodland	
Spotted Harrier	*Circus assimilis*	Open country	
Peregrine Falcon	*Falco peregrinus*	Open country	
Brown Falcon	*Falco berigora*	Open country	
Nankeen Kestrel	*Falco cenchroides*	Open country	
Peaceful Dove	*Geopelia placida*	Woodland	
Common Bronzewing	*Phaps chalcoptera*	Woodland	Yes
Crested Pigeon	*Ocyphaps lophotes*	Open country	
Galah	*Cacatua roseicapilla*	Open country	
Rainbow Lorikeet	*Trichoglossus haematodus*	Woodland	
Musk Lorikeet	*Glossopsitta concinna*	Woodland	
Purple-crowned Lorikeet	*Glossopsitta porphyrocephala*	Woodland	
Crimson Rosella	*Platycercus elegans*	Woodland	
Australian Ringneck	*Barnardius zonarius*	Woodland	
Red-rumped Parrot	*Psephotus haematonotus*	Woodland	Yes
Blue Bonnet	*Northiella haematogaster*	Woodland	
Elegant Parrot	*Neophema elegans*	Woodland	
Fan-tailed Cuckoo	*Cacomantis flabelliformis*	Woodland	Yes
Horsfield's Bronze-cuckoo	*Chrysococcyx basalis*	Woodland	
Rainbow Bee-eater	*Merops ornatus*	Woodland	
Varied Sittella	*Daphoenositta chrysoptera*	Woodland	Yes
Brown Treecreeper	*Climacteris picumnus*	Woodland	Yes
Superb Fairy-wren	*Malurus cyaneus*	Woodland	
Variegated Fairy-wren	*Malurus lamberti*	Woodland	
Spotted Pardalote	*Pardalotus punctatus*	Woodland	
Striated Pardalote	*Pardalotus striatus*	Woodland	
Weebill	*Smicrornis brevirostris*	Woodland	
Chestnut-rumped Thornbill	*Acanthiza uropygialis*	Woodland	Yes
Yellow Thornbill	*Acanthiza nana*	Woodland	Yes
Yellow-rumped Thornbill	*Acanthiza chrysorrhoa*	Woodland	Yes
Southern Whiteface	*Aphelocephala leucopsis*	Woodland	Yes
Red Wattlebird	*Anthochaera carunculata*	Woodland	
Little Wattlebird	*Anthochaera chrysoptera*	Woodland	
Spiny-cheeked Honeyeater	*Acanthagenys rufogularis*	Woodland	
Striped Honeyeater	*Plectorhyncha lanceolata*	Mallee Heath	
Singing Honeyeater	*Lichenostomus virescens*	Woodland	
Yellow-plumed Honeyeater	*Lichenostomus ornatus*	Open Mallee	
White-plumed Honeyeater	*Lichenostomus penicillataus*	Woodland	
Brown-headed Honeyeater	*Melithreptus brevirostris*	Woodland	Yes
White-naped Honeyeater	*Melithreptus lunatus*	Woodland	Yes
White-fronted Honeyeater	*Phylidonyris albifrons*	Mallee Heath	
New Holland Honeyeater	*Phylidonyris novaehollandiae*	Woodland	
Tawny-crowned Honeyeater	*Gliciphila melanops*	Mallee Heath	Yes
Eastern Spinebill	*Acanthorhynchus tenuirostris*	Woodland	Yes
White-browed Babbler	*Pomatostomus superciliosus*	Woodland	Yes
Red-capped Robin	*Petroica goodenovii*	Woodland	
Hooded Robin	*Melanodryas cucullata*	Woodland	Yes
Jacky Winter	*Microeca fascinans*	Woodland	Yes
Southern Scrub-robin	*Drymodes brunneopygia*	Mallee Heath	
Grey Shrike-thrush	*Colluricincla harmonica*	Woodland	
Golden Whistler	*Pachycephala pectoralis*	Woodland	
Rufous Whistler	*Pachycephala rufiventris*	Woodland	Yes
Grey Fantail	*Rhipidura fuliginosa*	Woodland	
Willie Wagtail	*Rhipidura leucophrys*	Woodland	Yes
Restless Flycatcher	*Myiagra inquieta*	Woodland	Yes
Australian Magpie-lark	*Grallina cyanoleuca*	Open country	
Black-faced Cuckoo-shrike	*Coracina novaehollandiae*	Woodland	
White-winged Triller	*Lalage tricolor*	Woodland	
Dusky Woodswallow	*Artamus cyanopterus*	Woodland	Yes
Grey Butcherbird	*Cracticus torquatus*	Woodland	
Australian Magpie	*Gymnorhina tibicen*	Open country	
Grey Currawong	*Strepera versicolor*	Woodland	
Little Raven	*Corvus mellori*	Open country	
White-winged Chough	*Corcorax melanorhamphos*	Woodland	Yes
Welcome Swallow	*Hirundo neoxena*	Open country	
Richard's Pipit	*Anthus novaeseelandiae*	Open country	
Skylark	*Alauda arvensis*	Open country	
European Goldfinch	*Carduelis carduelis*	Open country	
Zebra Finch	*Taeniopygia guttata*	Woodland	Yes
Red-browed Finch	*Neochima temporalis*	Woodland	Yes
Diamond Firetail	*Stagonopleura guttata*	Woodland	Yes
Mistletoebird	*Dicaeum hirundinaceum*	Woodland	
Silvereye	*Zosterops lateralis*	Woodland	Yes
Common Starling	*Sturnus vulgaris*	Open country	

Appendix 2. List of the 19 declining species considered for analysis in Chapter 2, with the number of cells in which they were present (Cells; of 22), ranges in their relative abundance (Range), type of woodland remnant used for the Woodland Remnant landscape variable (Woodland Remnant) and the remnant vegetation types used to calculate Aggregation (Aggregation Remnant types). Relative abundances are the summed total of all birds recorded across the two surveys and do not reflect actual bird numbers. For Woodland Remnant, 'All Woodland' refers to a combination of all the remnant types considered as both Open Woodland and Shrubby Woodland in Appendix 4. In Aggregation Remnant types, 'ALL' refers to all of the remnant vegetation types, and '-' minus the following types. Floristic details of the remnant types can be found in Appendix 4.

Species	*Cells*	*Range*	*Woodland Remnant*	*Aggregation Remnant types*
Brown-headed Honeyeater	22	4 - 106	All Woodland	ALL - Heath & Shrubland
Brown Treecreeper	7	0 - 5	Open Woodland	Open Woodland
Common Bronzewing	22	2 - 39	All Woodland	ALL - Heath
Diamond Firetail	20	0 - 55	Open Woodland	Open Woodland
Dusky Woodswallow	16	0 - 23	Open Woodland	Open Woodland
Hooded Robin	19	0 - 19	Open Woodland	Open Woodland
Jacky Winter	14	0 - 11	Open Woodland	Open Woodland
Red-capped Robin	19	0 - 21	All Woodland	All Woodland
Restless Flycatcher	12	0 - 7	Open Woodland	Open Woodland
Red-rumped Parrot	19	0 - 55	Open Woodland	Open Woodland
Rufous Whistler	17	0 - 9	All Woodland	All Woodland
Silvereye	18	0 - 30	All Woodland	ALL
Southern Whiteface	12	0 - 48	Open Woodland	Open Woodland
Varied Sittella	15	0 - 22	All Woodland	ALL - Heath & Shrubland
White-browed Babbler	22	24 - 198	All Woodland	ALL - Heath
Willie Wagtail	22	5 - 34	Open Woodland	Open Woodland
White-winged Chough	21	0 - 285	Open Woodland	Open Woodland
Yellow-rumped Thornbill	22	7 - 111	Open Woodland	Open Woodland
Yellow Thornbill	22	2 - 51	All Woodland	ALL - Heath

Appendix 3. Correlation matrix of the landscape variables according to Pearson's r. Bold indicates r values > 0.8 or < -0.8. Only Aggregation of All Vegetation is shown for Aggregation, as the alternative Aggregation variables had similar relationships with the other landscape variables.

	RV	All Rem	Wood Rem	OW Rem	Prox Rem	Prox Wood	Prox OW	Ajoin Rem	Ajoin Wood	Ajoin OW	RV Size	RV Shape	Tot Size	Tot Shape	Aggregation	Grazed pc	Grazed RV	Drainage	Rainfall	Diversity	Allocas	Callitris
All Rem	-0.46																					
Wood Rem	-0.36	0.80																				
OW Rem	-0.31	0.63	**0.90**																			
Prox Rem	-0.25	0.70	0.76	0.73																		
Prox Wood	-0.23	0.57	0.79	0.78	**0.94**																	
Prox OW	-0.05	0.30	0.55	0.73	0.77	**0.86**																
Ajoin Rem	-0.29	**0.82**	**0.96**	**0.89**	**0.80**	0.78	0.58															
Ajoin Wood	-0.28	0.68	**0.95**	**0.94**	0.75	**0.84**	0.68	**0.93**														
Ajoin OW	-0.15	0.42	0.73	**0.92**	0.61	0.73	**0.82**	0.76	**0.88**													
RV Size	0.77	-0.49	-0.51	-0.51	-0.27	-0.31	-0.24	-0.47	-0.48	-0.44												
RV Shape	-0.46	0.44	0.60	0.65	0.22	0.31	0.25	0.52	0.61	0.57	-0.55											
Tot Size	0.47	0.11	0.09	0.13	0.50	0.35	0.37	0.21	0.09	0.10	0.59	-0.24										
Tot Shape	-0.40	0.76	0.79	**0.80**	0.71	0.63	0.53	**0.84**	0.74	0.67	-0.54	0.69	0.23									
Aggregation	0.35	0.08	-0.05	0.00	0.26	0.15	0.24	0.12	0.03	0.10	0.37	-0.19	0.56	0.10								
Grazed pc	-0.16	0.08	0.40	0.51	0.20	0.34	0.43	0.37	0.44	0.54	-0.44	0.43	-0.13	0.44	-0.10							
Grazed RV	0.41	-0.13	0.16	0.28	0.00	0.13	0.32	0.19	0.26	0.42	0.00	0.18	0.07	0.17	0.09	0.80						
Drainage	0.19	-0.03	0.23	0.40	-0.05	0.09	0.24	0.26	0.38	0.56	-0.10	0.53	-0.04	0.31	0.03	0.52	0.62					
Rainfall	0.21	-0.69	-0.45	-0.18	-0.30	-0.17	0.12	-0.49	-0.31	0.00	0.33	-0.19	0.07	-0.42	0.11	0.19	0.18	0.15				
Diversity	-0.47	0.58	0.69	0.74	0.42	0.42	0.39	0.69	0.64	0.62	-0.68	0.60	-0.14	0.71	-0.31	0.67	0.40	0.36	-0.30			
Allocas	0.25	0.19	0.37	0.56	0.37	0.43	0.63	0.44	0.52	0.69	-0.08	0.52	0.32	0.54	0.35	0.37	0.51	0.67	0.03	0.30		
Callitris	0.23	-0.01	0.14	0.30	-0.01	0.07	0.28	0.17	0.26	0.43	-0.09	0.20	-0.01	0.04	-0.10	0.07	0.29	0.55	0.04	0.27	0.45	
OEW	-0.18	0.09	0.29	0.46	0.39	0.47	0.65	0.28	0.36	0.50	-0.24	-0.04	0.02	0.09	0.03	0.27	0.15	-0.19	0.19	0.32	0.05	0.09

Key: RV = Revegetation, All Rem = All Remnant, Wood Rem = Woodland Remnant, OW Rem = Open Woodland Remnant, Prox Rem = Proximity All Remnant, Prox Wood = Proximity Woodland Remnant, Prox OW = Proximity Open Woodland Remnant, Ajoin Rem = Ajoining Remnant, Ajoin Wood = Ajoining Woodland Remnant, Ajoin OW = Ajoining Open Woodland Remnant, RV Size = Revegetation Patch Size, RV Shape = Revegetation Patch Shape, Tot Size = Total Patch Size, Tot Shape = Total Patch Shape, Aggregation = Aggregation All Vegetation, Grazed pc = % Grazed Revegetation, Grazed RV = Grazed Revegetation, Drainage = Drainage Length, Rainfall = Average Annual Rainfall, Diversity = Habitat Diversity, Allocas = Proximity Allocasuarina, Callitris = Proximity Callitris, OEW = Proximity Open Eucalypt Woodland.

Appendix 4. Key to the **r**emnant vegetation types in Chapter 2 used to classify the remnant variables, generate the Habitat Diversity variable and discern the relevant habitat to include in the Aggregation variable. The components are the major vegetation types as a function of the dominant plant species that were grouped to form each remnant type. Woodland groups indicate which type of woodland used for the Woodland Remnant variable (Open Woodland or All Woodland) the remnant type corresponds, which cross references with the relevant woodland types for each individual bird species in Appendix 2. All Woodland comprises all the Open Woodland and Shrubby Woodland remnant types.

Remnant type	*Components*	*Woodland group*
Gum Woodland	*Eucalyptus camaldulensis, E. fasciculosa, E. leucoxylon*	Open Woodland
Box Woodland Open	*Eucalyptus porosa, E. odorata*	Open Woodland
Box Woodland Shrubby	*Eucalyptus porosa over mixed shrubs*	Shrubby Woodland
Allocasuarina Woodland	*Allocasuarina verticillata*	Open Woodland
Callitris Woodland	*Callitris gracilis var. preissii*	Open Woodland
Open Mallee	*Eucalyptus phenax, E.oleosa, E.gracilis, E. calycogona, E.incrassata (grazed)*	Open Woodland
Shrubby Mallee	*Eucalyptus incrassata, E.socialis, E. leptophylla over mixed shrubs*	Non Woodland
Heath	*Babbingtonia beyerii, Leptospermum coriacieum, Hibbertia australis, Glischrocaryon behrii*	Non Woodland
Shrubland	*Acacia paradoxa, A. pycnantha, A. rhigiophylla, Melaleuca acuminata, M. lanceolata, M. uncinata*	Non Woodland

Appendix 5. Number of parameters (k) and explained deviance (D^2) for the global models of each response variable in Chapter 2. The global models listed do not include the extra quadratic variables tested for some responses, as these were all unimportant and not included in final models (see Chapter 2 for details).

Response	*k*	D^2
Woodland Species	9	0.69
Brown-headed Honeyeater	8	0.74
Brown Treecreeper	9	0.68
Common Bronzewing	8	0.65
Diamond Firetail	9	0.80
Dusky Woodswallow	8	0.47
Hooded Robin	8	0.33
Jacky Winter	8	0.75
Red-capped Robin	9	0.52
Restless Flycatcher	8	0.63
Red-rumped Parrot	8	0.43
Rufous Whistler	8	0.42
Silvereye	8	0.40
Southern Whiteface	9	0.89
Varied Sittella	8	0.57
White-browed Babbler	8	0.80
Willie Wagtail	8	0.79
White-winged Chough	8	0.80
Yellow-rumped Thornbill	8	0.88
Yellow Thornbill	9	0.59

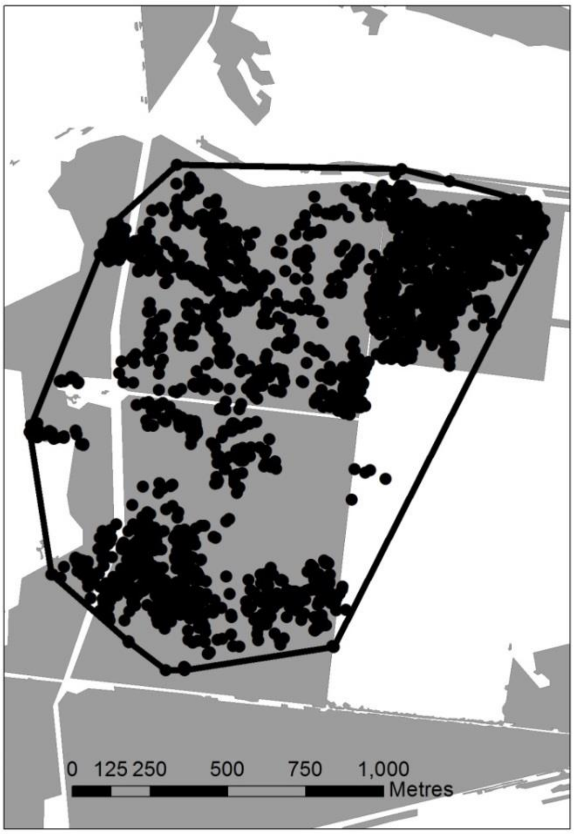
Appendix 6. Historical records and average abundances of the 19 declining species in a patch of previously surveyed revegetation compared to that recorded for the corresponding 2 x 2 km cell in Chapter 2. The historical data was based on 27 area search surveys conducted over eight years. Records indicate the number of surveys in which a species was recorded (of 27 for the historical data and of two for the current). Note that the amount of revegetation was greater in the current data (119 ha versus 63 ha) and so the comparisons are only general.

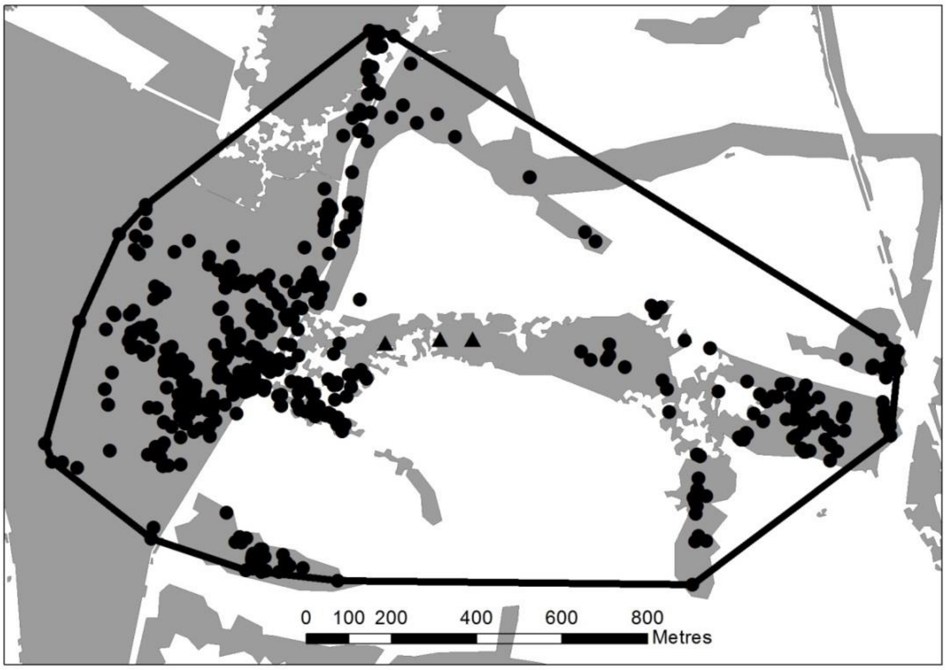
	Historical		*Current*	
Species	*Records*	*Average*	*Records*	*Average*
Brown-headed Honeyeater	25	20.7	2	38.5
Common Bronzewing	15	1.6	2	7.5
Diamond Firetail	14	2.1	2	3.5
Dusky Woodswallow	17	3.6	2	7.5
Hooded Robin	23	3.3	2	9.5
Jacky Winter	3	0.2	0	0.0
Red-capped Robin	27	6.7	2	6.5
Red-rumped Parrot	0	0.0	2	4.5
Restless Flycatcher	10	0.6	2	1.5
Rufous Whistler	14	0.8	1	3.0
Silvereye	23	7.1	2	11.5
Varied Sittella	8	1.6	0	0.0
White-browed Babbler	27	21.9	2	36.0
White-winged Chough	23	12.1	2	5.0
Willie Wagtail	18	1.9	2	4.0
Yellow Thornbill	26	9.2	2	14.0
Yellow-rumped Thornbill	26	14.6	2	21.5

Appendix 7. List of nine woodland species that were detected in historical surveys of a patch of revegetation, but undetected in the corresponding 2 x 2 km cell of Chapter 2. Also listed are the number of times they were recorded (of 27 total surveys) and their movement status based on experience of these species in the region.

Species	*Records*	*Movement status*
Black-faced Cuckoo-shrike	2	Nomadic
Chestnut-rumped Thornbill	6	Sedentary
Jacky Winter	3	Sedentary
Musk Lorikeet	5	Nomadic
Tree Martin	3	Summer migrant
Varied Sittella	8	Sedentary
White-naped Honeyeater	9	Winter migrant
White-winged Triller	1	Summer migrant
Yellow-faced Honeyeater	1	Winter migrant

Appendix 8. Examples of area used within the three largest home ranges from the three species tracked in chapter 3. Top left = Varied Sittella group VS_gmyy, top right = Restless Flycatcher group RF_omby, and bottom = Brown-headed honeyeater group BHH_gmyb. Minimum Convex Polygon (MCP) boundaries are shown as solid black lines, locations where birds were recorded as black dots, and vegetated areas in grey. Triangle shaped dots represent records on inaccessible land around which usage could not be established. For the corresponding group statistics, see Appendix 9.





Appendix 9. Individual group statistics for each of the 85 home ranges considered in Chapter 3. Groups are listed as a combination of the colour band or band number of the main bird followed, or group of colour banded birds at a particular site. Site locations and descriptions can be found in the corresponding studies (see Table 3.1) and in Chapter 4. Area statistics are: Total Area = Entire area of home range including non-vegetated areas, Veg Area = Summed area of all vegetation (woodland) in home range, % RV = Percentage of vegetation area made up of revegetation. Patches = number of patches used in the home range. Species (Spp) codes are listed in alphabetical order, and described below. Continued over page.

Spp	Site	Group	Records	Days	Years	Total Area	Veg Area	%RV	Patches
BHH	RVB	BHH_gmyb	581	12	2	172.7	77.0	91.8	9
BHH	RV2	BHH_wmwn	701	8	1	101.3	61.7	99.8	5
BHH	RV3	BHH_wmyw	1229	12	1	70.4	46.1	100.0	7
BHH	RV4	RV4_BHH_1	713	20	5	39.5	26.4	97.3	7
BTC	RV1	RV1_BTC_1	519	19	1	13.8	11.5	37.9	1
BTC	RVB	RVB_BTC	1108	14	1	10.2	9.3	80.4	3
BTC	RV1	BTC_bmgy	79	6	1	9.0	7.4	51.1	2
CRT	MZ1	MZ1_CRT_2	783	16	1	17.8	16.0	0.0	2
CRT	MZ1	MZ1_CRT_1	862	13	1	16.2	15.3	0.0	1
CRT	RV4	RV4_CRT_1	30	3	2	11.6	11.6	99.6	1
CRT	RV4	CRT_YMBG	113	3	1	9.1	8.7	41.7	1
DF	RVB	DF_rmon	77	8	1	30.7	22.9	28.4	5
DF	RVB	DF_rmoy	80	8	1	21.5	14.9	8.1	5
DF	RVB	DF_rmwb	54	8	2	13.9	11.0	41.2	3
DF	RVB	DF_rmyb	22	3	1	8.0	5.9	10.9	4
DF	RVB	DF_rmyy	31	4	1	5.6	2.0	1.7	3
GOW	RV4	GOW_ymwr	15	9	3	18.4	18.4	100.0	1
HR	RV4	RV4_HR_3	179	23	4	48.3	36.6	81.4	1
HR	RV4	RV4_HR_1	1529	46	5	31.5	31.5	90.9	1
HR	RV4	RV4_HR_2	1121	14	2	24.9	24.5	84.5	1
HR	RV1	RV1_HR_1	68	9	4	14.4	13.7	90.2	1
HR	RV4	HR_bmro	76	20	3	12.2	12.2	99.8	1
HR	RVB	RVB_HR_1	63	13	1	9.4	8.8	75.9	3
HR	MZ1	MZ1_HR_1	96	14	2	13.5	8.7	0.0	2
ONJ	RV4	ONJ_052-10254	37	10	1	129.1	66.1	54.2	8
ONJ	RV4	ONJ_052-11165	44	6	1	42.8	39.6	100.0	3
ONJ	RV5	ONJ_052-11170	38	8	1	16.8	16.7	99.1	2
ONJ	RV4	ONJ_052-10805	62	17	1	18.5	12.9	92.3	1
ONJ	RV4	ONJ_052-10252	82	21	1	11.7	11.0	100.0	2
ONJ	RV4	ONJ_052-10804	28	11	1	15.3	11.0	98.5	1
ONJ	RVB	ONJ_052-11162	29	6	1	11.4	10.9	100.0	2
ONJ	RV4	ONJ_052-11166	44	8	1	13.6	8.8	80.3	1
ONJ	RV3	ONJ_052-10260	32	3	1	8.7	8.7	100.0	1
ONJ	RV5	ONJ_052-11172	23	6	1	3.8	3.8	100.0	1
ONJ	RVB	ONJ_052-11161	29	6	1	2.2	2.1	98.2	2
RCR	RV4	RV4_RCR_1	1048	23	2	9.5	9.5	90.6	1
RCR	RV4	RV4_RCR_2	1222	18	2	8.7	8.7	94.1	1
RCR	RV4	RV4_RCR_5	58	23	4	7.0	7.0	78.5	1
RCR	RV4	RV4_RCR_3	59	16	3	6.5	6.5	100.0	1
RCR	RV1	RCR_wmgg	357	8	2	6.0	5.4	99.6	1
RCR	RV4	RV4_RCR_4	54	10	3	5.3	5.3	100.0	1
RCR	RV1	RCR_wmoo	624	8	1	6.0	4.2	94.4	1
RCR	RV4	RCR_wmrr	21	10	3	4.1	4.1	69.9	1

Appendix 9. *(Continued)*

Spp	Site	Group	Records	Days	Years	Total Area	Veg Area	%RV	Patches
RCR	RV1	RCR_wmwg	21	3	2	3.3	2.6	100.0	1
RCR	RV4	RCR_bmnb	26	6	2	1.4	1.4	99.8	1
RCR	RV4	RCR_wmoy	28	6	1	1.1	1.1	100.0	1
RF	RV2	RF_omby	2934	14	1	188.0	156.8	100.0	5
RUW	Rocky Gully	RUW_rmoo	1111	9	1	132.2	90.6	3.0	6
RUW	RV4	RV4_RUW_1	2043	15	2	76.5	54.7	73.5	2
RUW	RVB	RUW_rmow	655	7	1	27.1	24.7	99.5	3
RUW	RVB	RVB_RUW_1	606	5	1	23.5	20.0	63.8	4
RUW	RVB	RVB_RUW_2	904	11	2	19.7	19.7	99.3	1
SFW	RV4	SFW_wmwy	23	9	4	8.2	8.2	100.0	1
SFW	RV1	RV1_SFW_1	152	12	5	5.4	5.4	89.4	1
SFW	RV4	SFW_wmrg	20	6	3	4.8	4.8	97.9	1
SWF	Frahn's Farm	SWF_Grp3	945	9	1	22.4	16.9	3.3	1
SWF	Wattle Road	SWF_Grp4	1106	8	1	19.6	14.0	7.0	2
SWF	Frahn's Farm	SWF_Grp5	605	8	1	12.6	10.2	3.3	1
SWF	Frahn Lane	SWF_Grp1	1265	11	1	30.1	9.8	66.1	2
SWF	Frahn Lane	SWF_Grp6	560	6	1	13.7	6.5	99.5	5
SWF	Frahn Lane	SWF_Grp2	1117	9	1	14.1	6.2	85.2	2
VFW	RV1	VFW_gmbo	38	6	4	9.5	9.5	90.1	1
VS	RVB	VS_gmyy	2009	19	1	246.3	165.8	85.3	10
VS	RV3	VS_gmwo	1226	15	1	160.1	134.2	97.7	8
VS	RV2	VS_gmgb	1523	9	1	73.4	67.9	99.2	4
WBB	RV4	WBB_gmrw	38	11	4	38.5	38.1	94.2	2
WBB	RV4	WBB_gmrr	35	13	5	21.7	21.7	100.0	1
WBB	RV4	WBB_wmyy	28	6	5	18.8	18.8	100.0	1
WBB	RV4	WBB_gmoo	38	13	4	15.7	15.7	97.6	1
WBB	RV4	WBB_gmyy	65	17	4	15.5	15.3	97.7	2
WBB	RV4	WBB_gmbw	21	11	4	13.4	13.4	100.0	1
WBB	RV4	WBB_gmbb	22	8	4	13.1	13.1	100.0	1
WBB	RV4	WBB_gmyr	33	9	4	11.3	11.2	100.0	2
WBB	RV4	WBB_gmog	30	6	4	6.6	6.6	79.9	1
WBB	RV4	WBB_gmow	23	8	5	6.5	6.5	73.7	1
WBB	RV4	WBB_gmyg	35	5	3	4.1	4.1	100.0	1
YRT	RV4	YRT_BMRY	21	6	5	43.7	35.2	94.4	1
YRT	RV4	YRT_BMYY	38	8	3	29.8	29.5	98.9	1
YRT	RV4	YRT_bmgo	30	9	5	10.0	10.0	100.0	1
YRT	RV4	YRT_BMRW	84	12	3	8.8	8.7	100.0	1
YRT	RV4	YRT_BMRR	39	7	2	7.9	7.9	100.0	1
YRT	RV4	YRT_BMGR	31	11	4	7.7	7.7	100.0	1
YRT	RV4	YRT_YMRG	33	3	1	6.6	6.6	100.0	1
YRT	RV4	YRT_BMYG	21	3	2	3.3	3.3	100.0	1
YRT	RV4	YRT_BMMR	40	3	1	8.7	1.7	100.0	3

Species (Spp) code definitions: BHH = Brown-headed Honeyeater, BTC = Brown Treecreeper, CRT = Chestnut-rumped Thornbill, DF = Diamond Firetail, GOW = Golden Whistler, HR = Hooded Robin, ONJ = Owlet-nightjar, RCR = Red-capped Robin, RF = Restless Flycatcher, RUW = Rufous Whistler, SFW = Superb Fairy-wren, SWF = Southern Whiteface, VFW = Variegated Fairy-wren, VS = Varied Sittella, WBB = White-browed Babbler, YRT = Yellow-rumped Thornbill.

Appendix 10. 62 bird species observed in three or more surveys in at least one of the five survey sites in Chapter 4, with their broad habitat categorisation, and declining status in the Mount Lofty Ranges according to Paton *et al.* (2004). Species are listed in taxonomic order.

Common Name	Species Name	Broad Habitat	Declining?
Emu	*Dromaius novaehollandiae*	Open country	
Painted Button-quail	*Turnix varia*	Woodland	Yes
Brown Goshawk	*Accipiter fasciatus*	Woodland	
Collared Sparrowhawk	*Accipiter cirrhocephalus*	Woodland	
Brown Falcon	*Falco berigora*	Open country	
Peaceful Dove	*Geopelia placida*	Woodland	
Common Bronzewing	*Phaps chalcoptera*	Woodland	Yes
Crested Pigeon	*Ocyphaps lophotes*	Woodland	
Galah	*Cacatua roseicapilla*	Open country	
Rainbow Lorikeet	*Trichoglossus haematodus*	Woodland	
Musk Lorikeet	*Glossopsitta concinna*	Woodland	
Purple-crowned Lorikeet	*Glossopsitta porphyrocephala*	Woodland	
Crimson Rosella	*Platycercus elegans*	Woodland	
Australian Ringneck	*Barnardius zonarius*	Woodland	
Red-rumped Parrot	*Psephotus haematonotus*	Woodland	Yes
Horsfield's Bronze-cuckoo	*Chrysococcyx basalis*	Woodland	
Rainbow Bee-eater	*Merops ornatus*	Woodland	
Varied Sittella	*Daphoenositta chrysoptera*	Woodland	Yes
Brown Treecreeper	*Climacteris picumnus*	Woodland	
Superb Fairy-wren	*Malurus cyaneus*	Woodland	
Variegated Fairy-wren	*Malurus lamberti*	Woodland	
Spotted Pardalote	*Pardalotus punctatus*	Woodland	
Striated Pardalote	*Pardalotus striatus*	Woodland	
Weebill	*Smicrornis brevirostris*	Woodland	
Chestnut-rumped Thornbill	*Acanthiza uropygialis*	Woodland	Yes
Yellow Thornbill	*Acanthiza nana*	Woodland	Yes
Yellow-rumped Thornbill	*Acanthiza chrysorrhoa*	Woodland	Yes
Southern Whiteface	*Aphelocephala leucopsis*	Woodland	Yes
Red Wattlebird	*Anthochaera carunculata*	Woodland	
Spiny-cheeked Honeyeater	*Acanthagenys rufogularis*	Woodland	
Singing Honeyeater	*Lichenostomus virescens*	Woodland	
Yellow-plumed Honeyeater	*Lichenostomus ornatus*	Woodland	
White-plumed Honeyeater	*Lichenostomus penicillataus*	Woodland	
Brown-headed Honeyeater	*Melithreptus brevirostris*	Woodland	Yes
White-naped Honeyeater	*Melithreptus lunatus*	Woodland	Yes
New Holland Honeyeater	*Phylidonyris novaehollandiae*	Woodland	
White-browed Babbler	*Pomatostomus superciliosus*	Woodland	Yes
Red-capped Robin	*Petroica goodenovii*	Woodland	
Hooded Robin	*Melanodryas cucullata*	Woodland	Yes
Jacky Winter	*Microeca fascinans*	Woodland	Yes
Grey Shrike-thrush	*Colluricincla harmonica*	Woodland	
Golden Whistler	*Pachycephala pectoralis*	Woodland	
Rufous Whistler	*Pachycephala rufiventris*	Woodland	Yes
Grey Fantail	*Rhipidura fuliginosa*	Woodland	
Willie Wagtail	*Rhipidura leucophrys*	Woodland	Yes
Restless Flycatcher	*Myiagra inquieta*	Woodland	Yes
Australian Magpie-lark	*Grallina cyanoleuca*	Open country	
White-winged Triller	*Lalage tricolor*	Woodland	
Masked Woodswallow	*Artamus personatus*	Woodland	
White-browed Woodswallow	*Artamus superciliosus*	Woodland	
Dusky Woodswallow	*Artamus cyanopterus*	Woodland	Yes
Australian Magpie	*Gymnorhina tibicen*	Open country	
Grey Currawong	*Strepera versicolor*	Woodland	
Little Raven	*Corvus mellori*	Open country	
White-winged Chough	*Corcorax melanorhamphos*	Woodland	Yes
Welcome Swallow	*Hirundo neoxena*	Woodland	
Tree Martin	*Petrochelidon nigricans*	Woodland	Yes
House Sparrow	*Passer domesticus*	Open country	
Diamond Firetail	*Stagonopleura guttata*	Woodland	Yes
Mistletoebird	*Dicaeum hirundinaceum*	Woodland	
Silvereye	*Zosterops lateralis*	Woodland	Yes
Common Starling	*Sturnus vulgaris*	Open country	

Appendix 11. Correlation matrix of the microhabitat variables according to Pearson's r for a) the All Species dataset and b) the Declining Species dataset. Bold indicates r values > ± 0.5. Continued over page.

a)

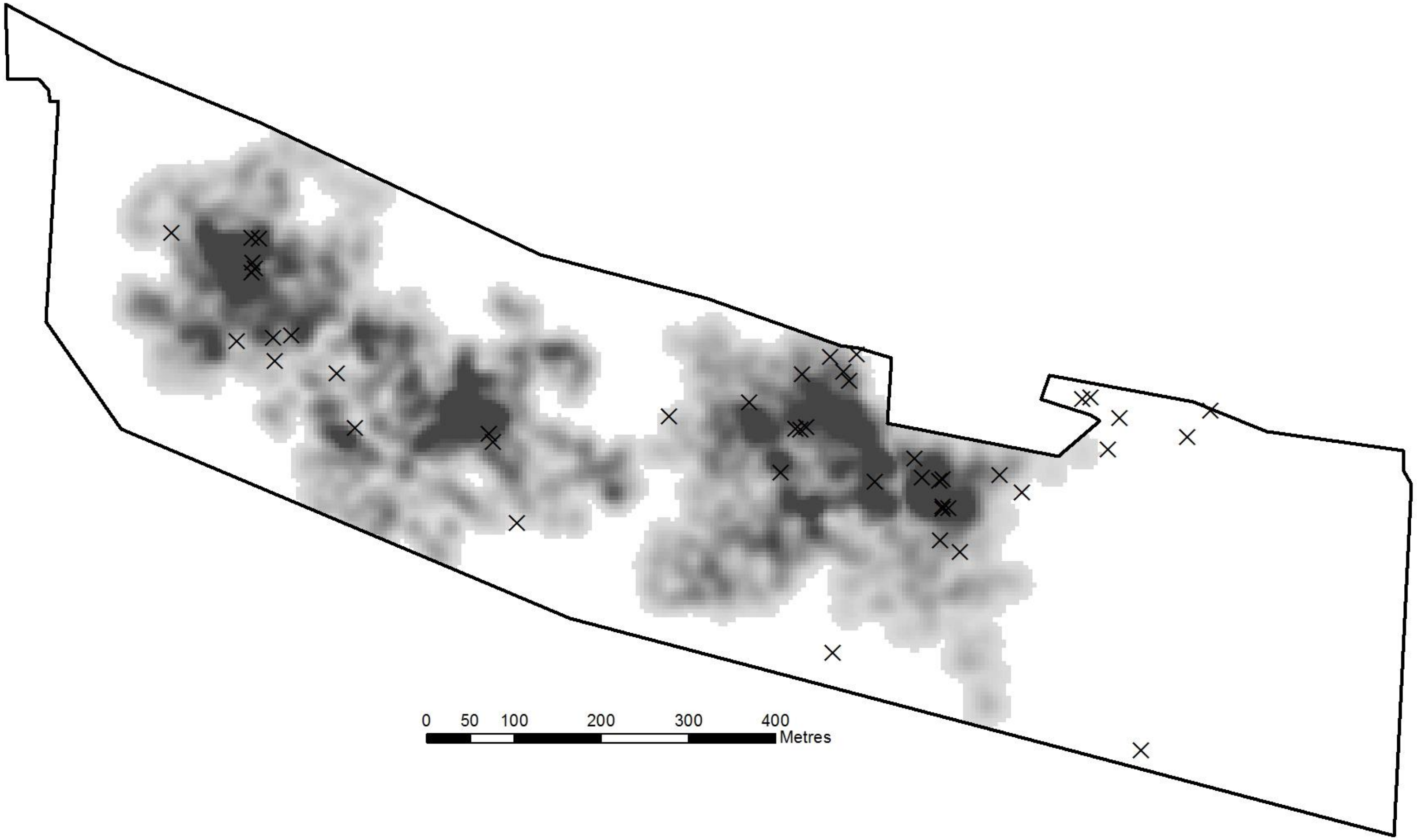
	Understorey	Overstorey	UnderOverH	FallenDead	DeadTrees	Bare	Litter	Moss	Grass	FallenTimber	SmallShrubs	GroundH	PlantDensity	UnderstoreyH	OverstoreyH	LocalPlants	ANN Plants	ANN Understorey	DistWoodRem
Overstorey	**-0.58**																		
UnderOverH	0.32	-0.44																	
FallenDead	0.03	-0.28	0.19																
DeadTrees	-0.11	-0.11	0.04	0.01															
Bare	-0.44	0.29	-0.09	0.11	0.05														
Litter	-0.11	0.46	-0.37	-0.36	0.01	-0.35													
Moss	-0.01	-0.34	0.43	0.23	0.06	0.04	**-0.61**												
Grass	**0.54**	-0.44	0.07	0.02	-0.05	-0.36	-0.37	-0.22											
FallenTimber	-0.18	-0.03	0.10	0.40	-0.01	0.00	-0.20	0.20	-0.12										
SmallShrubs	0.13	-0.41	0.38	0.32	-0.19	-0.01	**-0.56**	0.14	0.38	0.08									
GroundH	0.24	**-0.54**	0.44	0.37	-0.05	0.12	**-0.90**	0.44	**0.51**	0.25	**0.71**								
PlantDensity	-0.14	**0.73**	-0.38	-0.33	0.11	0.04	**0.57**	-0.39	-0.28	-0.21	-0.49	**-0.62**							
UnderstoreyH	0.32	-0.35	0.40	0.15	-0.14	-0.03	-0.12	0.16	-0.02	0.05	0.13	0.10	-0.21						
OverstoreyH	-0.53	**0.55**	-0.04	-0.13	-0.07	0.34	0.12	0.03	-0.42	0.03	-0.17	-0.25	0.32	-0.14					
LocalPlants	-0.22	**0.50**	-0.07	-0.12	-0.06	0.28	-0.05	-0.01	-0.10	-0.10	-0.01	0.02	0.32	-0.35	0.26				
ANN Plants	-0.43	0.35	-0.12	0.01	0.03	0.08	0.29	-0.06	-0.43	0.04	-0.12	-0.32	0.44	-0.04	0.18	0.19			
ANN Understorey	0.08	0.03	-0.09	0.11	0.13	-0.15	0.04	-0.09	0.17	0.03	-0.06	0.05	0.19	-0.11	-0.21	-0.03	0.08		
DistWoodRem	**0.60**	-0.30	-0.07	-0.19	0.02	**-0.52**	0.32	-0.22	0.28	-0.17	-0.26	-0.24	0.06	0.20	**-0.51**	**-0.53**	-0.29	0.14	
DistGrazedRV	-0.41	0.25	-0.36	-0.14	0.16	0.43	-0.01	0.29	**-0.50**	-0.09	**-0.50**	-0.26	0.14	-0.07	0.23	0.00	0.05	-0.14	-0.13

Appendix 11. *(Continued)*

b)

	Understorey	Overstorey	UnderOverH	FallenDead	DeadTrees	Bare	Litter	Moss	Grass	FallenTimber	SmallShrubs	GroundH	PlantDensity	UnderstoreyH	OverstoreyH	LocalPlants	ANN Plants	ANN Understorey	DistWoodRem
Overstorey	**-0.68**																		
UnderOverH	0.28	-0.45																	
FallenDead	0.09	-0.28	0.21																
DeadTrees	-0.19	0.01	-0.03	-0.01															
Bare	-0.26	-0.01	0.28	0.17	0.31														
Litter	-0.28	0.71	-0.59	-0.44	-0.05	-0.40													
Moss	0.22	-0.54	0.55	0.26	0.04	0.15	**-0.73**												
Grass	**0.51**	-0.50	0.01	0.15	-0.17	-0.43	-0.31	-0.11											
FallenTimber	0.04	-0.25	0.20	0.53	0.01	0.09	-0.39	0.17	0.08										
SmallShrubs	0.10	-0.38	0.35	0.21	-0.34	-0.06	**-0.54**	0.29	0.36	0.24									
GroundH	0.33	**-0.73**	0.58	0.38	0.00	0.24	**-0.94**	0.63	**0.41**	0.44	**0.68**								
PlantDensity	-0.39	**0.79**	-0.48	-0.27	0.20	-0.07	**0.71**	-0.49	-0.42	-0.20	-0.61	**-0.73**							
UnderstoreyH	0.28	-0.44	0.30	0.11	-0.10	0.13	-0.41	0.41	0.05	0.28	0.14	0.35	-0.38						
OverstoreyH	-0.55	**0.54**	0.03	-0.09	-0.08	0.35	0.19	-0.05	-0.56	-0.05	-0.09	-0.26	0.33	-0.12					
LocalPlants	-0.26	**0.44**	0.05	-0.13	0.16	0.11	0.09	0.09	-0.35	-0.27	-0.08	-0.07	0.36	-0.30	0.21				
ANN Plants	-0.51	0.45	-0.12	-0.24	0.31	-0.05	0.31	-0.09	-0.27	-0.27	-0.28	-0.34	0.53	-0.22	0.25	0.36			
ANN Understorey	-0.20	0.29	-0.14	-0.13	0.21	-0.15	0.29	-0.25	-0.01	-0.05	-0.36	-0.28	0.40	-0.07	0.01	0.17	0.35		
DistWoodRem	**0.57**	-0.29	-0.20	-0.18	-0.04	**-0.49**	0.31	-0.23	0.34	-0.15	-0.28	-0.25	0.01	0.07	**-0.57**	**-0.51**	-0.25	0.10	
DistGrazedRV	-0.42	0.23	-0.21	-0.09	0.36	0.41	0.09	0.04	**-0.44**	-0.21	**-0.54**	-0.31	0.28	-0.10	0.16	0.21	0.17	0.15	-0.17

Appendix 12. Comparison between the distribution of Red-capped Robins at site RV4 from systematic area search mapping used in Chapter 4 (crosses) to that obtained in a study targeting this species during the same period (grey to black shading; Northeast 2007). The surface shown was adapted from data collected in Northeast (2007) and is a composite of the kernel density surfaces for four pairs of Red-capped Robins, where darker values indicate more frequently used areas. Note that the crosses east of the kernel distribution are in an area that was not surveyed for this species in the targeted study, hence the lack of overlap in this area.



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